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Restoration Ecology of Cattle Grazing on Mixed Prairie Wellsites:
A Hierarchical Approach

by

Etienne Martin Joseph Soulodre ©

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

Department of Renewable Resources

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Faculty of Graduate Studies and Research

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled *Restoration Ecology of Cattle Grazing on Mixed Prairie Wellsites: A Hierarchical Approach* submitted by Etienne Soulodre in partial fulfillment of the requirements for the degree of Master of Science.

Dedication

This thesis is dedicated to my parents who taught me to love the outdoors and to work hard

Abstract

A hierarchical approach was used to study revegetation of petroleum wellsites in native Mixed Grass Prairie in southeastern Alberta. Three seeding treatments, natural recovery (unseeded) and undisturbed prairie were compared from 1996 to 2000. In 1999 and 2000 grazing treatments were applied to three year old wellsites and evaluated at population, community, ecosystem and landscape scales. For complete erosion control, native wheatgrass cultivars provided an effective, rapid ground cover. However, native wheatgrass cultivars, even in small percentages of a seed mix, resulted in wheatgrass communities with low species diversity that did not return to predisturbance conditions. Cattle grazing curbed aggressive wheatgrasses. Natural Recovery was an effective long-term strategy for reestablishing the predisturbance plant community although erosion risk is higher than for seeded treatments. Cattle grazing was not an appropriate management strategy for Natural Recovery wellsites as it impeded community development and ecosystem function.

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CHAPTER I RESTORATION AS A HIERARCHICAL PROCESS

1.0 The Theoretical and Practical Roots of Restoration

As human impact on the environment has become a social and intellectual focus in the twentieth century, various movements have emerged with potential responses/solutions to face the challenges of environmental degradation. While most approaches have appropriately focused on the prevention of human impacts through changing resource use patterns (Berry 1966) or protecting remnant areas from impact (Ehrenfield 1970), an alternate approach has been to attempt to repair the damage inflicted by humans to the environment (Berger 1985). Such ameliorating approaches are generally taken when sudden destructive events (e.g. oil spills) requiring immediate remediation occur. However, repair of more chronic disturbances such as mining lands and abandoned cultivation has also been attempted. The "repair" of disturbed lands can generally be grouped under the heading of "restoration" (Hobbs and Norton 1996). The specific meaning of this word and the extent of its use varies and is widely contested. These meanings will be discussed further in an attempt to create a unifying framework for scientifically studying restoration.

The North American roots of restoration can be traced to early prairie restorations conducted at the University of Wisconsin-Madison Arbortoreum (Jordan et al. 1987). These early restoration efforts consisted of planting prairie grasses into abandoned cropland and subsequently conducting prescribed burns. Since these first efforts in the 1930s restoration has slowly evolved as a practice. In the fields of prairie conservation, mine reclamation and wetland reconstruction, significant practical advances were made and restoration practitioners learned how to make

rough approximations of the systems they wished to recreate (Jordan et al. 1988, Berger 1985). It took until the 1980s however, for restoration to be considered more than "gardening with native species" (Allen and Hoekstra 1992).

Beginning in the 1980s, restoration practitioners began to attempt to formalize both the foundations of their activities and the ways in which restoration research was conducted (Jordan et al. 1987). Ecological principles such as succession or ecosystem function began to inform the way restoration practitioners viewed their craft. Restoration ecology was not only to prove the usefulness of ecology but was to be the new frontier for fundamental research. In an entire volume dedicated to the subject, eminent ecologists from across the field professed faith that restoration projects would provide the "acid test" of ecology (Bradshaw 1987). The manipulation of ecological systems provided by restoration projects was to provide research opportunities that otherwise would be too expensive or immoral. The way to prove that ecology understands ecosystems was to try and put one back together (Harper 1987). A new marriage of research and restoration was proposed.

While much progress has been made in the last ten years in restoration, most notably in the creation of journals dedicated to the subject, the research-practice renaissance envisioned by the contributors to the Jordan et al. (1987) publication has not taken place. The restoration literature still consists primarily of documenting site specific cases of what works or what doesn't (Hobbs and Norton 1996) or proposing conceptual frameworks for further work. The most important achievement of restoration in the last ten years may have been to garner a wider audience of interest and support.

The wide interest in restoration has facilitated attempts to develop an *a priori* foundation for restoration ecology. Current debates in the restoration literature center around definitions of restoration (Jackson et al. 1995), measures of restoration success (Aronson et al. 1993) and approaches to restoration research (Majer and Recher 1994). The theoretical foundations of restoration itself have yet to mature.

The development of a mature restoration ecology will depend on the creation of conceptual frameworks for research and practice (Hobbs and Norton 1996). Most people interested in pursuing the intellectual foundations of restoration will also be practitioners. This allows a unique opportunity for restoration to develop a pragmatic intellectual base and avoid the growing pains of other disciplines where it often takes years for theory and practice to find a common reference point.

2.0 The Semantics of Restoration

2.1 Outline for organizing definitions of restoration

For restoration to develop as a discipline, a clear intellectual foundation is required. Definitions must be more than operational. Language is powerful and has meaning and so will influence how activities are conducted or understood (Higgs 1997). Convention is not a good enough reason to accept definitions and seemingly trivial discussions are necessary before substantial progress can be made.

So far in this chapter, restoration has been used in the broadest sense to refer to any attempt to ameliorate the effect of human disturbance on the environment. This includes activities from very local attempts to reestablish plant communities to the regional attempts to realign the human economy with the environment. Definitions of restoration vary from the extremely narrow to the extremely general.

Narrow definitions have the advantage of being clear and specific while the scope is limited. General definitions, while having a broader scope, usually are vague. The following section will attempt to summarize the major types of restoration definitions from narrow to broad.

2.2 Restoration as replicating predisturbance conditions

In the strictest sense, restoration is the process of returning the environment to the exact conditions that existed prior to human disturbance (Jordan et al. 1988). The ecosystem must be exactly replicated in composition, structure and function. This type of restoration is sometimes defined in such rigid terms that it is presented as an unattainable goal. However, most practitioners use this meaning of the word in a more pragmatic sense (restoration *sensu latu*) (Aronson et al. 1993). This type of restoration attempts to reestablish an approximation of the plant and animal communities/populations that existed before human disturbance. This approach is most common among pure ecologists, especially plant ecologists. This definition of restoration is tied to a focus on plant and animal communities/populations. The success of restoration is evaluated by the degree to which the restored plant communities/populations represent predisturbance condition.

Within the ranks of those who consider restoration the establishment of plant and animal communities there has been recent debate on how exactly restoration success should be evaluated. Recent disagreements include defining what constitutes natural or predisturbance conditions. For example, native plant communities are dynamic through time and shift in composition (Jackson et al. 1995). The question then becomes what properly constitutes a predisturbance plant community. For example, some plant communities may simply be relicts of post glaciation plant

migration. It may not be appropriate to recreate plant communities that are only a temporary feature of the landscape and have not adjusted to any kind of environmental equilibrium (Sprugel 1991). Biological invasions present a similar problem. Exotic plant and animal species often invade and become naturalized as part of the native plant communities. Should these plants be considered part of the plant community that the restorationist is trying to create? To take the problem a step further, we can question whether a “natural” state of plant or animal communities really exists (Hobbs and Norton 1996).

Many plant/animal communities in North America prior to European settlement were heavily influenced by the cultural practices of the Amerindians. In large parts of Europe, “native” landscapes have not existed for over a millennium. Native plant communities have been replaced by “cultural landscapes” that are closely tied to the agricultural production systems of local communities (Higgs 1997, Naveh 1994). Restorationists have in these cases argued that establishment of these cultural landscapes may be the definition of restoration which is most appropriate. Even in this case of a broader conception of what constitutes restoration, the emphasis is still on plant and animal communities/populations.

2.3 Restoration as reestablishing ecosystem function

Some restorationists have proposed that rather than focus on plant/animal assemblages the goal of restoration is to reestablish ecosystem function (Bradshaw 1987). Ecosystem processes such as nutrient cycling, energy flow and stability are emphasized (Ehrenfield and Toth 1997). Practitioners who work with drastically disturbed lands have usually advocated this approach (Bradshaw 1983). For example in mined lands where little soil is present, or substrate has limiting factors such as

toxic metals, practitioners are content to establish a perennial plant cover that is "stable and self-sustaining" (NRC 1992). Rather than frame their practices in terms of community succession, the approach of this school has been to view restoration in terms of ecosystem development (*sensu* Odum 1967). In North America especially, this type of ecosystem repair has been distinguished from other forms of restoration by calling it reclamation (Jordan et al. 1988). Aronson et al. (1993) suggested "rehabilitation" as a term for the returning of productivity (a component of ecosystem function) to the land. The important contribution of this definition of restoration has been to focus attention at the larger scale of the ecosystem.

2.4 Restoration as preservation of the environment

As interest in restoration has grown, so has the number of professionals insisting that what they do is restoration. Many different activities that contribute to the preservation of natural environments are considered restoration (Hobbs and Norton 1996). For example, prescribed burning of a prairie remnant, reintroducing extirpated birds, controlled flooding of wetlands and managed harvesting of a natural resource can all be included under the title of restoration (Berger 1985). A recent but now defunct definition by the Society for Ecological Restoration stated that anything that preserves "ecosystem health" is restoration (Higgs 1997). With such a wide definition, it is hard to conceive of many conservation activities that would not constitute restoration. The advantage of this broad definition is that it focuses the attention of restorationists on the landscape scale. All activities that contribute to the overall conservation of undisturbed land on the landscape must be considered. It is no longer sufficient to focus on the reconstruction of a single piece of habitat. We

must consider the reconstruction and preservation of the landscape as a whole (Naveh 1994).

2.5 Restoration as reestablishment of productive uses

While restoration has traditionally focused on ameliorating human disturbance by returning the land to some natural or semi-natural cultural state, some practitioners have argued that conservation is only one of many human values that restored land could fulfill (Hobbs and Norton 1996). Disturbed land could be returned to many other productive human uses such as cropland, urban landscapes, artificial wetlands or completely artificial landscapes. This process is called "reallocation" by Aronson et al. (1993). In the province of Alberta, this idea is expressed legislatively in the term "equivalent capability" (Powter 2000). Restoration (reclamation in the language of the legislation) can be considered a success if the use of the restored land has an equivalent capability as the predisturbance land even if the land use is completely different. For example, a gravel pit could be turned into a dug out and because the dug out is considered as useful as the predisturbance pasture, then the restoration is considered successful.

As another example, often the purpose of mine tailing restoration is not to establish a functioning ecosystem nor to reassemble plant communities but rather to provide a final useful landscape. For example, some oilsands reclamation in the Fort McMurray region seeks to reestablish bison pasture where boreal forest previously existed. The significance of this approach is that it tries to focus our attention on the social value placed in landscapes that may not have anything to do with the conservation of natural environments. Some restorationists have argued that restoration often needs to fulfill aesthetic values (Turner 1992). Obviously this is an

example of focusing restoration on the human dimension of landscapes rather than any intrinsic ecosystem features.

2.6 Restoration as a process of social transformation

Some restorationists have argued for an even wider scope for restoration. Restoration can be seen as social progress. No longer just a medium for society to achieve certain social goals, restoration is a unique act that changes the nature of human relationships with the environment. Western civilization no longer only exploits the natural environment but also has a moral obligation to repair it. Fidelity to a certain concept of pristine nature or kind of productive landscape is immaterial.

Early restorationists recognized that there was something socially unique about their activity. Prairie restorations conducted by the University of Wisconsin-Madison Arboretum were " the first time anyone ever tried to put a complex plant community back together" (Berger 1985). Higgs (1997) argued that restoration represents a process of social rejuvenation whereby society becomes redemocratized and realigned with ecological values. Alternately, restoration has been called an "expensive self-indulgence for the upper classes" (Kirby 1994) or a process of "transvaluing shame into beauty" (Turner 1992). From the wider sociological perspective, the meaning and success of restoration hinges not on the technical questions of concern to ecologists, but rather on the degree to which restoration creates social change.

3.0 Hierarchy of Scales

3.1 Restoration as a hierarchy of scales

The previous section organized definitions of restoration by the spatial and temporal scales that they focus on. Definitions of restoration that focus on reestablishing some form of natural plant community are usually at the scale of either population or plant community. Definitions that focus on reestablishing ecosystem function focus at the ecosystem scale. Definitions that emphasize conservation focus on the landscape scale. When restoration is considered as a process to change a socially desirable goal for a piece of land, then restoration is focusing on the human social level which occurs at a larger scale above individual landscapes. When restoration is seen as a process of social transformation, the scale even supersedes social values and exists at a suprasocial scale.

For an ecologist, organizing definitions of restoration along a continuum of spatial and temporal scales is useful for a number of reasons. First, it allows the ecologist to have multiple working definitions of what constitutes restoration and to organize these multiple definitions in an intelligible manner. Secondly, each scale of definition corresponds to a natural scale of the ecosystem and this hierarchy of scales also provides a useful framework for understanding ecosystems from a purely empirical perspective.

3.2 Ecological systems as a hierarchy of scales

It has been proposed that ecosystems may be best understood as a hierarchy of scales (Allen and Starr 1982). Population, community, ecosystem and landscape processes occur at increasing spatial and temporal scales. Hierarchy theory is an approach to ecology that tries to recognize the importance of this hierarchy of scales.

Allen and Hoekstra (1992) argued that when scientists investigate an ecosystem they must be aware that their investigation has an extent, the largest entity observed, and a grain, the smallest entity observed. By explicitly recognizing that the ecosystem processes we observe are scale dependent we can avoid erroneously extrapolating processes from one scale to another. The best way to understand ecosystems may be to study them across a range of ecological scales and then try and determine how the various scales are linked (Allen and Starr 1982). Natural systems need to be studied across the population, community, ecosystem and landscape scales.

While the importance of scale has been recognized (Allen and Hoekstra 1987), restoration has not been studied from an explicitly hierarchical perspective. A hierarchical perspective allows a "concordance of scales" between restoration definitions and natural scales of the ecologist. If such an approach is useful for the study of restoration (restoration ecology), then it should advance both the fundamental understanding of the restored system and produce practical management recommendations. This thesis investigates the revegetation of petroleum wellsites in native Mixed Grass Prairie (MGP) in a hierarchical framework.

MGP is a diminishing resource and so effective restoration techniques are needed. A hierarchical approach may help produce these needed restoration techniques. Investigations were conducted at the population, community, ecosystem and landscape scales. Investigations across these scales are then combined to gain a more complete understanding of the restoration process and produce practical management recommendations. Using the revegetation of MGP wellsites as a case study, this thesis tests the assertion that such a hierarchical approach is useful for restoration ecology.

4.0 Model System – Revegetation of Mixed Grass Prairie Wellsites

4.1 Mixed grass prairie

The semiarid grasslands of the Northern Great Plains of North America are called the mixed grass prairie (MGP). MGP in the drier parts of the Canadian prairies on loamy Chernozemic soils was described by Coupland (1961) as the *Stipa-Bouteloua* and *Bouteloua-Stipa* faciations; the latter occurring in the driest parts of the region. Vegetation on Solonetzic soils in this area is classified as the *Bouteleoua-Agropyron* faciation. These faciations are approximately equivalent to the Dry-Mixed Grass subregion of Strong and Leggat (1992).

MGP continues to disappear at an alarming rate. Coupland (1987) concluded that between 1956 and 1981, 19% of the remaining MGP was destroyed. As a native ecosystem, the remaining MGP has high conservation value. Continuing threats to MGP include industrial development, livestock mismanagement and cultivation (Coupland 1987).

4.2 Wellsite revegetation in the mixed grass prairie

MGP is under threat from oil and gas wellsite development. Over 30,000 active and 10,000 abandoned wells were found in this region of Alberta in 1991 (Kerr et al. 1993). Well site construction usually involves the destruction of vegetation and stockpiling of topsoil and subsoil on a 100 x 100 m area (Kerr et al. 1993). Wellsite revegetation in Alberta is regulated by reclamation criteria used for certification (AEP 1995). Certification involves inspection by provincial government officers to guarantee meeting of minimum soil and vegetation criteria.

Revegetation practices for MGP wellsites prior to 1992, usually consisted of seeding agronomic species such as *Agropyron pectiniforme* R. & S. (crested

wheatgrass). These species have been recognized as inadequate for wellsite revegetation as they prevent the establishment of native prairie, invade the surrounding areas and present a grazing management problem (Gerling et al. 1996). In recent years there has been an increased emphasis on the use of native plants for revegetation in Alberta (Gerling et al. 1996). Current revegetation of wellsites in MGP typically utilizes a mixture of *Agropyron dasystachyum* (Hook.) Scribn. (northern wheatgrass), *Agropyron trachycaulum* (Link) Malte (slender wheatgrass), and *Agropyron smithii* Rydb. (western wheatgrass), which represent a significant improvement over the use of *Agropyron pectiniforme* (Gerling et al. 1996).

4.4 Revegetation alternatives: native prairie revegetation research project

Increased emphasis on conservation of native prairie led researchers to search for more effective revegetation techniques. For example, it has been suggested that more species rich seed mixes may better reestablish the prairie ecosystem (Hammermeister 2001). Alternatively, not seeding (natural recovery) and allowing secondary succession to take place may be a better revegetation technique. A third proposed management practice is to introduce cattle grazing to the sites earlier in the revegetation process. The necessity of managing restoration projects has been widely recognized (Wark et al. 1996) and cattle grazing presents an obvious management option for MGP wellsites.

To investigate these alternative revegetation practices researchers, government and industry created the Native Prairie Revegetation Project. The project examined increasing the species richness of seed mixes and using natural recovery (no seeding) as alternative revegetation techniques. This thesis research was conducted as part of the Native Prairie Revegetation Project Dry Mixed Grass Prairie subproject. The

research project was established in 1996 and Hammermeister (2001) completed a Ph.D. dissertation on the first three years of the project. In 1999 this M.Sc. thesis research was initiated and grazing treatments were established and continued in 2000. Chapters II and III of this thesis focus on the population scale to investigate seed bank and demographic dynamics from data collected in 1999 and 2000. Chapters IV and V focus on the community and ecosystem scales to investigate successional trends and ecosystem function using data from 1996 to 2000. Chapter VI focuses on the landscape scale to investigate cattle patch grazing selection on revegetated wellsites using data from 1999 and 2000. Chapter VII synthesizes information across all of the scales investigated and presents management recommendations for the revegetation of MGP wellsites.

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Chapter II Soil Seed Bank and Mixed Grass Prairie Wellsite Revegetation

1.0 Introduction

The soil seed bank, the germinable seeds in the upper few cm of soil, is an important component of grassland ecology. Seed banks contribute to understanding succession (Peart 1989a), species interactions (Peart 1989b) and recovery after disturbance. The important role of the seed bank in the revegetation of anthropogenically disturbed lands has been demonstrated in tropical pastures (Tsuyazaki and Kanda 1996), British heathlands (Putwain and Gillham 1990), northern tundra (Ebersole 1989), coal mines in southern Saskatchewan (Archibald 1980) and North Dakota (Iverson and Wali 1982). Seed bank ecology may also be useful for improving revegetation practices of disturbed grasslands.

Seed banks may provide an essential source of desired plant propagules in revegetation but also act as a source of undesired weeds. However, little research has focused on improving revegetation practices using seed bank ecology. Additionally, revegetation practitioners often use the predisturbance ecosystem as the measuring stick for reclamation success, thereby implying a goal to establish a soil seed bank similar to predisturbance. This unstated goal might be prerequisite for the site to be self sustaining and stable (Bradshaw 1987).

Revegetation practices may alter site vegetation by manipulating available plant propagules in disturbed systems. This is usually done by seeding desirable species. Mowing or spraying for weed control together with grazing and fertilizing

also influence the available seed rain on disturbed land (Figure 2-1). Off site seed rain provides an additional source of propagules (Iverson and Wali 1982).

Available seed rain may be altered before it enters the seed bank by vegetation that influences microclimate and light, and therefore induces dormancy (Williams 1983), affects lateral seed movement, seed predation (Chambers and Macmahon 1994) and seed-soil contact. Grazing may impact seed rain by mechanically pushing seed into the soil. After seed rain filtering, propagules may enter either the persistent or transient seed bank (Thompson and Grime 1979). Seeds in the transient seed bank are available for recruitment for a short period of time while those in the persistent seed bank remain available over many years through dormancy. The persistent seed bank therefore accumulates from past events and can be influenced by the extent that disturbed soils carry the “memory” of the predisturbance seed bank (Templeton and Levin 1979).

Oil wellsite construction is a common disturbance on Mixed Grass Prairie (MGP) in southern Alberta (Kerr et al. 1993). Typical revegetation practices include seeding with native grasses and excluding grazing for four to five years to ensure recovery. Alternatively, allowing natural (unseeded) secondary succession to take its course may be a better approach to reestablish the native ecosystem (Hammermeister and Naeth 1996). Grazing has been proposed as an alternative management strategy for wellsites early in the revegetation process. The necessity of managing restoration projects has been widely recognized (Wark et al. 1996) and cattle grazing presents an obvious management option for MGP wellsites. All of these management practices (natural recovery, seeding, grazing) have the potential to impact the soil seed bank and thus, revegetation success.

The objective of this research was to investigate the MGP germinable soil seed bank in seeded, unseeded and native prairie treatments with and without grazing four years after disturbance. Seed bank ecology will then be used to help make recommendations for wellsite revegetation and management in MGP.

2.0 Methods

2.1 Site description

The study took place in four native prairie sites in southeastern Alberta, Canada from May 1999 to August 2000 (Figure A-1). Two sites were located 300 m apart, approximately 20 km northwest of Medicine Hat on loam textured Orthic Brown Chernozemic soils. The remaining two sites were 3 km apart, 15 km northeast of Bow Island on sandy loam textured Orthic Brown Chernozemic soils. Natural vegetation in this area is referred to as dry mixed grass prairie (Strong and Leggat 1992) or MGP (Coupland 1961). Undisturbed prairie at each of the sites was dominated by *Stipa comata* Trin & Rupr. (needle and thread) and *Bouteloua gracilis* (HBK) Lag. (blue grama). Taxonomy follows Moss (1983). The climate is semiarid and the majority of growing season precipitation falls in June. May to September precipitation was slightly below long-term averages in 1996 to 1997 and moderately above long-term averages in 1999 (Table A-1). In 2000 precipitation was approximately half the long-term averages.

Each of the research sites was a drilled and abandoned wellsite located inside a field of native prairie used for grazing. Cattle have grazed each of the fields for over thirty years. In 1999, the range condition of the sites was estimated to vary from good to excellent (Wroe et al. 1988).

Wellsites were established in either fall 1995 or spring 1996 and abandoned in spring 1996. Each of the companies contributing wellsites was asked to reclaim them using normal practices up to the point of being seeded. Each wellsite had the topsoil and subsoil separately removed and stockpiled. After industrial activities were finished, topsoil and subsoil were replaced and a seedbed prepared. Each wellsite and an adjacent strip of undisturbed native prairie were fenced to exclude grazing for the first three years of the study.

2.2 Treatments and experimental design

Three seeded and one non-seeded (Natural Recovery) revegetation treatments were compared to the Undisturbed treatment in this study. The seeded treatments consisted of a three species, wheatgrass dominated mix (Current), a five species mix (Simple) and a 21 species mix (Diverse) (Table A-2). Each of these treatments was investigated with and without cattle grazing.

Each wellsite was treated as a block. The three seeded and Natural Recovery treatments were placed in the same location within each block because of the influence of predominant wind direction on natural revegetation (Table A-2). In contrast, position of the Undisturbed treatment varied with each block. Exclosures allowed comparison of Ungrazed and Grazed treatments within each revegetation treatment. The experiment was treated as a strip plot design. For this experiment, only Grazed and Ungrazed Natural Recovery, Undisturbed and Simple treatments were investigated.

2.3 Treatment application

Revegetation treatments were implemented in spring 1996 by Hammermeister (2001). Seeding rates (kg/ha) were calculated based on a desired number of pure live

seeds (PLS) per unit area (300 PLS/m^2). Because purity and viability data were not available for the forbs, they were estimated as 66%. Chick starter was used as a carrier to ensure even seed distribution and improve seed flow through the drill openers. The total seeding rate was applied with two perpendicular passes of a calibrated Truax native seed drill. *Koeleria macrantha* (Ledeb.) J.A. Schultes f. (june grass) was broadcast with chick starter as a carrier following seeding. After seeding a straw crimp was applied to seeded and Natural Recovery treatments by spreading straw bales and mechanically crimping straw into the soil. Although the seeder was calibrated, the seeding rate varied slightly due to limited accuracy of openers (10%). The species seeded in the simple treatment were *Agropyron smithii* Rydb (western wheatgrass), *Agropyron dasystachyum* (Hook) Scribn (northern wheatgrass), *Stipa comata*, *Koeleria macrantha* and *Bouteloua gracilis*.

During summer 1999 grazing exclosures were built in the center of each site enclosing part of each revegetation treatment (Figure A-2). The unfenced portion of each treatment was grazed in June and July 1999. Grazing treatments were applied by moving cattle from the adjoining field into the fenced wellsite where the animals were free to graze any of the treatments outside the exclosure. Water was provided in troughs placed directly beside the gate of each wellsite. Cattle remained in the wellsite until visually estimated utilization approached 30 to 50% on each treatment. A confounding factor to the experimental design is that two of the sites (Int East and Int West) were mowed two weeks prior to the application of the grazing treatment in 1999. Because of this complication, stocking rates of the Grazing treatment were reduced for these two sites (Table A-3). In summer 2000, mowed and unmowed sites were visually similar and all blocks were included in the analysis.

2.4 Soil seedbank sampling

The soil seed bank was sampled in each subplot by taking 20, 3.8 cm diameter cores to a 6 cm depth with a plunger sampler from June 19 to 22, 2000. Sampling locations were random within each revegetation/grazing treatment combination subplot. All samples from each subplot were composited and stored in a plastic bag until June 23. Sampling was conducted through any ground cover such as litter or live vegetation to not disturb seeds in the upper cm of soil. Soil samples were spread on plastic trays 20 cm wide by 50 cm long on a 2 to 3 cm layer of vermiculite and larger bits of live vegetation and debris were removed. From June 23 to August 28, trays were kept in the header house of a greenhouse, randomly placed on two benches. Ambient daytime temperatures ranged between 20 and 27 ^{o}C. No additional lighting was provided. Trays were watered as necessary to keep them moist, usually once a day. Seedlings were counted every two weeks. Following identification, individual seedlings were removed from the tray. The small number of seedlings not identifiable were called unknown forbs.

2.5 Data analyses

Species were classified as either graminoids (perennial grasses and sedges) or weeds (annual grasses and dicots). Seed density was calculated as number of seeds/m^2 based on the total area sampled by the 20 cores. Data were analyzed using analysis of variance (ANOVA). The experiment was treated as a split plot design (Appendix III). Treatment means were compared using the least significant difference (LSD) at alpha equal to 0.05. SAS software was used to analyze the data using Proc GLM (SAS Institute Inc. 2000). Proc GLM uses the wrong error terms for split designs and so correct F-tests were obtained by using a random statement and

specifying the error term in mean comparison tests. Inspection of mean sums of squares revealed that in some cases the residual error was larger than the block by treatment interactions (Error terms A and B). To find correct error terms Proc Mixed was used to calculate F-tests and conduct mean comparisons.

Given the small sample size, tests were not made for normality or homogeneity of variance. ANOVA is generally regarded as robust to violations of both of these assumptions (Day and Quinn 1989). The third assumption of ANOVA, independence of samples, is also violated in this study by the lack of randomization at the treatment level. The fixed location of treatments in this study increases the risk of a Type I error. Many precedents justify this analysis in field studies however, and choice of an analysis should reflect the judgment and objectives of the researcher rather than inflexible statistical rules of thumb (Stewart-Oaten 1995).

3.0 Results and Discussion

3.1 General results

A total of 202 seedlings of 23 species were identified in all 24 trays. *Agropyron trachycaulum* (Link) Malte. (slender wheatgrass) was the most abundant graminoid although none were found in the Undisturbed treatment (Table 2-1). *Salsola kali* L. (russian thistle), the most abundant weed, was also in low abundance in the Undisturbed control. In the Natural Recovery treatment, *Koeleria macrantha* was the next most common grass while *Kochia scoparia* (L.) Schrad. (kochia weed) was the next most common weed. In the Undisturbed treatment, *Carex* spp. (sedges) was the most abundant graminoid while unknown forbs were the most abundant

weed. In the Simple treatment *Carex* spp. was the second most abundant graminoid while *Sonchus* spp. (sow thistle) was the second most common weed.

Significant effects were found for grazing and revegetation treatment on number of weed seedlings (Table 2-2). Nonsignificant effects were found for block ($p = 0.57$), block by grazing interactions ($p = 0.89$) and revegetation treatment by block interactions ($p = 0.11$). Because of near significant grazing by treatment interactions ($p = 0.09$) least squares means were investigated. Grazing increased weed seed density only in the Natural Recovery and decreased it in the Undisturbed treatment (Table 2-3). The Natural Recovery had more weed seedlings than the Simple and Undisturbed treatments only when grazed.

Nonsignificant effects for number of graminiod seeds were found for block ($p = 0.20$), grazing ($p = 0.13$), revegetation treatment ($p = 0.26$), block by grazing interaction ($p = 0.76$) and block by revegetation treatment interactions ($p=0.31$). Near significant effects were found for revegetation treatment by grazing interaction ($p= 0.09$) and so least squares means were investigated. When grazed, the Natural Recovery treatment had significantly fewer graminoid seedlings than the Simple treatment (Table 2-3). Grazing reduced number of grass seedlings in the Undisturbed treatment.

Significant effects for total number of seeds were found for revegetation treatments (Table 2-2) and the revegetation treatment by grazing interaction ($p= 0.04$). Nonsignificant effects were found for block ($p=0.99$), grazing ($p = 0.41$), block by grazing interaction ($p = 0.62$) and revegetation treatment by block interactions ($p=0.86$). Investigation of least means squared indicated that under the Grazing

treatment the Simple and Natural Recovery treatments had more total seedlings than the Undisturbed (Table 2-3).

3.2 Influence of extant vegetation on the seed bank

The germinable soil seed bank in this study did not reflect the composition of predominant vegetation, however, the revegetation treatments had important differences in their seed banks. The Natural Recovery treatment, dominated by annual weeds, had significantly more weeds in the seed bank than the Undisturbed treatment when grazed. This provides evidence that the extant vegetation may influence the composition of the seed bank by contributing to the seed rain rather than by influencing the incoming seed rain from both offsite and onsite sources. Most studies found that the germinable seed bank does not closely reflect the composition of the vegetation (Harper 1977, Coffin and Laurenroth 1989) however, extant vegetation does usually impact broad patterns in the seed bank (Peart 1989a).

Offsite seed rain was also important in this study. For example, the most abundant perennial grass that germinated was *Agropyron trachycaulum*. This species was not present in the Simple treatment but was seeded in the adjacent revegetation treatments (Diverse and Current). The most logical explanation for this is that the seed blew in from adjacent treatments.

The high amount of *Agropyron trachycaulum* in the Simple treatment may be explained by the vegetation influencing incoming seed rain. The Simple treatment had the highest above ground biomass (Chapter VI). Williams (1979) and Quinton and Willms (1995) argued that litter cover could significantly impact the seed bank by altering microclimate. *Agropyron trachycaulum* seeds may have dispersed into the

Undisturbed treatment, however, they may not have entered the germinable seed bank due to inadequate litter cover to influence dormancy or capture seeds.

3.3 Impact of "memory" on the seed bank

Since both the Natural Recovery and Simple treatments differ in composition from the Undisturbed, soil disturbance and revegetation has altered the persistent seed bank. However, there is evidence that seed bank "memory" is important. In 1998, the Simple treatment had a high annual forb component while in 1999 and 2000, few annuals persisted (Chapter V). *Salsola kali*, an annual weed, was present in large quantities in the seed bank of the Simple treatment in 2000. Seed produced by weeds in 1998 and 1997 therefore appear to persist in the seed bank after these species have disappeared from the plant community. In all treatments, several unidentified rosettes were found in the seed bank that were not part of the existing vegetation. These species likely were in the seed bank before disturbance and persisted through revegetation.

3.4 Impact of grazing on seed bank

In this study, grazing treatment significantly increased the density of weed seeds in the seed bank although this effect was driven by the Natural Recovery treatment alone. This would suggest that grazing may retard succession by increasing the number of weed seeds in the Natural Recovery treatment. Other researchers have found that grazing retards succession in the seedbank. Johnston et al. (1969) found, in both Fescue and MGP, that grazing decreased the number of grasses and increased the number of weeds in the seed bank. Willms and Quinton (1995) found that the total number of seeds increased with grazing pressure and that the increase in number

of seeds was accounted for primarily by an annual forb and an exotic grass. In contrast in this study, grazing reduced the number of weed seeds in the Undisturbed treatment. This would suggest that grazing of MGP prairie may actually improve the successional status of the seed bank.

3.5 Soil seedbank and revegetation strategies

Common measures of revegetation success often include "self-sustaining" as a criterion (Bradshaw 1987). For vegetation to be self-sustaining, it is necessary that propagules be added to the seed bank for future recruitment. Often undisturbed sites are used as references for measuring reclamation success. Revegetation practitioners may therefore consider soil seed bank replenishment to predisturbance conditions as a goal for "success".

At a coarse level of resolution this study presents evidence that four years after establishment, the seed bank of both the Ungrazed Natural Recovery and the Ungrazed and Grazed Simple treatment has become equivalent to that of predisturbance conditions in terms of number of germinable graminoid seeds. It is important for revegetation practitioners however, to look at the processes occurring in the seed bank rather than just the total number of desirable seeds.

Because of the high density of germinable graminoid seeds in both the Simple and Natural Recovery treatments, we may conclude that these treatments likely will add enough seed to the seed bank to ensure that the system will be self-sustaining. Martelette and Anderson (1986) came to similar conclusions about plantings of crested wheatgrass on a reclaimed mine. They argued that onsite vegetation was contributing enough to the seed bank for the community to be self-perpetuating. However, in the Simple treatment, the dominant grasses reproduce vegetatively.

Recruitment from the seed bank may not be important for the seeded community that is dominated by rhizomatous grasses.

In the Undisturbed and Natural Recovery treatments, important bunchgrass species such as *Stipa comata* can only recruit from the seed bank. Species such as *Hordeum jubatum* L. (foxtail barley) and *Koeleria macrantha* were found in the Natural Recovery and these seeds play an important role in the future development of the plant community. However, such perennial grass species do not usually form a persistent seed bank in grasslands and so must be recruited from the transient seed bank (Roberts 1986). This study may not have detected the seeds of species that are present in the seed bank during only a small part of the year. For example, abundant seed rain of *Stipa comata* was witnessed in the Natural Recovery treatment but none was found in the seed bank. *Stipa comata* has been establishing in the Natural Recovery and so must be establishing from the transient seed bank.

In this study, soil memory, soil disturbance, seeding, extant vegetation, offsite seed rain and grazing all apparently impacted the seed bank during revegetation. Practitioners need to be aware of the role of the seedbank in their system when designing management strategies. Both the Simple and Natural Recovery revegetation treatments in the MGP appear adequate to reestablish the predisturbance seed bank in terms of number of grass seeds. Grazing, however, will probably impede seed bank reestablishment in the Natural Recovery treatment.

3.6 Accuracy of the data

Although the ecology of northern MGP has been studied extensively, few studies have focused on the seed bank. The total densities of germinable seeds reported in this study (198 to 700/m^2) is similar to previous densities reported for

MGP by Johnston et al. (1969) (780/m^2) and Coffin and Laurenroth (1989) (122 to 2748/m^2). Higher densities of seeds are usually reported for more mesic North American grasslands such as Tallgrass Prairie (Rabinowitz 1981), (6470/m^2) and Fescue prairie (Willms and Quinton 1995) (1790 to 7783/m^2).

Two obstacles limiting the accuracy of soil seed bank studies have been methodological problems (Forcela 1984, Bigwood and Inouye 1988) and spatial heterogeneity of the seedbank. This study used a relatively large number of small cores distributed randomly to account for the spatial variability inherent in the seed bank. Sampling was not done through time however, and so the ability to detect the transient seed bank was hampered.

Many seeds could have been in the soil samples that did not germinate but were viable. The greenhouse conditions that the samples were subjected to may have acted as an environmental sieve (Harper 1977) activating germination in some species but not others. Johnston et al. (1969) found that temperature affected the germination of seeds from MGP soils in the greenhouse. Interpretation of data to make management recommendations in this study is probably best done combining statistical results with close inspection of the raw data.

4.0 Conclusions

Natural Recovery and grazing will increase the number of weed seeds in the seed bank of wellsite disturbances in MGP. Both a Simple (seeded) and Ungrazed Natural Recovery treatment will likely establish enough seeds in the seedbank for the plant community to be self sustaining. Grazing may retard plant community development in the Natural Recovery treatment and may improve it in the

Undisturbed. Many aspects of seed bank ecology including on and offsite seed rain, seed bank memory and persistent and transient seed banks were needed to explain the development of revegetation treatments in the MGP.

Table 2-1: Mean germinable seed density (seeds/m^2) by species for the soil seed bank from Grazed and Ungrazed revegetation treatments.

Species	Natural Recovery		Undisturbed		Simple	
	Grazed	Ungrazed	Grazed	Ungrazed	Grazed	Ungrazed
<u>Graminoids</u>						
Agropyron dasystachyum	0	0	0	0	22 (25)[1]	0
Agropyron trachycaulum	11 (22)	187 (293)	0	0	209 (204)	143 (177)
Bouteloua gracilis	0	0	0	11 (22)	0	0
Carex spp.	0	0	55 (55)	55 (83)	22 (25)	33 (42)
Hordeum jubatum	11 (22)	11 (22)	0	0	22 (44)	0
Koeleria macrantha	44 (88)	33 (66)	33 (42)	0	0	11 (22)
Poa spp.	0	0	0	11 (22)	0	0
Setaria glauca	0	11 (22)	0	0	0	0
Stipa comata	0	0	0	11 (35.5)	0	0
Stipa viridula	11 (22)	0	11 (22)	11 (22)	0	0
<u>Weeds</u>						
Amaranthus spp.	11 (22)	11 (22)	0	0	0	0
Artemisia frigida	22 (25)	11 (22)	11 (22)	0	0	0
Avena fatua	22 (44)	33 (66)	0	0	0	0
Chenopodium album	0	11 (22)	0	22 (44)	0	0
Descurania spp.	22 (25)	0	0	0	0	0
Kochia scoparia	55 (66)	33 (66)	11 (22)	0	0	22 (44)
Lappula spp.	11 (22)	11 (22)	0	11 (44)	11 (22)	0
Lepidium densiflorum	0	0	0	0	0	11 (22)
Unidentified forb (Mustard)	0	0	0	22 (25)	0	0
Unidentified forb (Rosette 1)	55 (110)	0	88 (80)	11 (22)	0	11 (22)
Unidentified forb (Rosette 2)	33 (66)	0	0	55 (110)	0	11 (22)
Unidentified forb (Rosette 3)	0	0	11 (22)	0	0	0
Salsola kali	385 (270)	154 (116)	0	22 (44)	176 (266)	55 (83)
Setaria viridis	0	11 (22)	0	0	0	0
Sonchus spp.	44 (88)	33 (42)	0	0	55 (66)	66 (76)

[1]Standard deviation in brackets

Table 2-2: Mean density of germinable seeds (seeds/m^2) from revegetation and grazing treatments in mixed grass prairie in 2000.

Treatment	Grass	Weeds	Total
Revegetation			
P-value	0.26	0.03	0.01
Standard Error	72	76	68
Natural Recovery	110a[1]	467a	578a
Simple	236a	209b	446b
Undisturbed	209a	121b	330b
Grazing			
P-value	0.17	0.04	0.37
Standard Error	42	47	55
Grazed	154a	323a	477a
Ungrazed	216a	209b	425a

[1]Means within a column with different letters are significantly different

Table 2-3: Least squares means of germinable graminoid, weed and total seeds (seeds/m^2) from Grazed and Ungrazed revegetation treatments in mixed grass prairie in 2000.

Revegetation Treatment	Standard Error	Grazed	Ungrazed
Grass			
Natural Recovery	89	77a[1](b)[2]	143a(a)
Simple	89	286a(a)	187a(a)
Undisturbed	89	99b(ab)	319a(a)
Weed			
Natural Recovery	122	627a(a)	308b(a)
Simple	122	242a(b)	176a(a)
Undisturbed	122	99a(b)	143b(a)
Total			
Natural Recovery	96	704a(a)	451b(a)
Simple	96	528a(a)	363a(a)
Undisturbed	96	198a(b)	462b(a)

[1]Means within a row with different letters are significantly different within a revegetation treatment
[2]Means within a column with different letters are significantly different within a grazing treatment

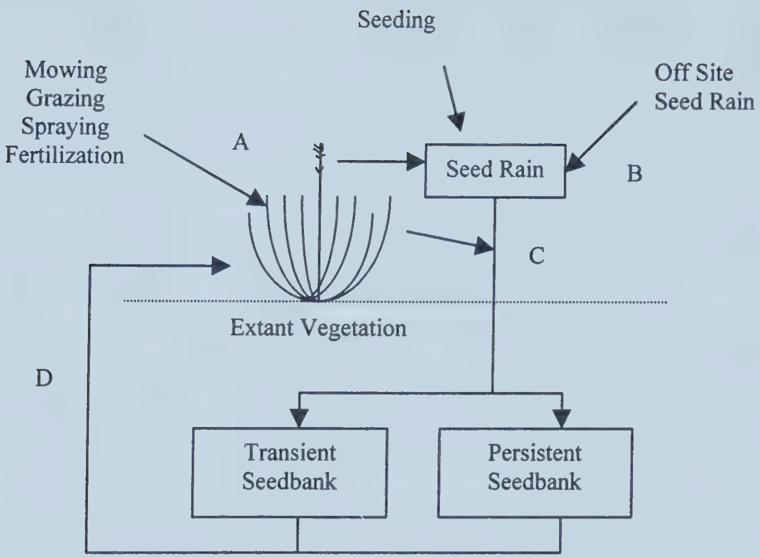

Figure 2-1: Role of the soil seed bank in revegetation. A - Management activities change composition, structure and seed production of vegetation. B - Seed rain provides pool of seeds to enter the seed bank. C - Vegetation acts as filter by influencing dormancy, allelopathy, predation and movement of seeds to enter seed bank. D - Seed bank provides propagules for establishment of vegetation.

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Chapter III Plant Population Processes and Mixed Grass Prairie Wellsite Revegetation

1.0 Introduction

Plant population biology has been identified as a discipline that may contribute to restoration ecology (Montalvo et al. 1997). One component of plant population biology, plant demography, the study of birth and death patterns in plants, may be especially useful for improving restoration practices. Restoration efforts often use rhizomatous grasses, the population patterns of which are poorly understood.

In plant population biology it has long been accepted that demography occurs as a hierarchy of parts in clonal species (Harper 1977). Birth and death events occur at the scale of leaf (phytomer), stem (tiller), plant or a larger cohort such as the genet (Hartnett and Bazzaz 1985c). Several tillers (ramets) may be physiologically integrated and therefore behave as a primary demographic unit instead of the single tiller (Bernard 1990). The genet (all members of a single clone) is ultimately the unit of interest in evolution (Hartnett and Bazzaz 1985c). Each of these scales may respond to different forces depending on the temporal scale of events that drive population processes. For example, leaf demography may depend on short-term physiological needs occurring over days while grazing events that occur on the scale of weeks may influence tillering. Most plant population studies examine changes within only a single scale (e.g. Zhang and Romo 1992). This can lead to erroneous conclusions especially in clonal plants where the definition of the important demographic unit may be arbitrary.

Plant population modeling has become sophisticated enough to accurately model neighborhood effects such as density dependent mortality and size hierarchies.

Despite the success of this modeling, it is dangerous to try and extrapolate these results to the scale of the plant community (Allen and Starr 1982). Community processes occur at larger spatial and temporal scales than neighborhood population processes. Neighborhood population models may not be able to explain the complex phenomena occurring at the community scale.

Confusion in ecology occurs when we fail to explicitly define the scale at which processes are occurring (Allen and Starr 1982). Scale has two components; extent, the largest entity observed, and grain, the smallest entity observed. Plant population processes may help explain patterns at the extent of plant community provided we focus on demography across the entire community rather than at the neighborhood scale. For example, at the neighborhood scale tiller mortality may be density dependent but occur at rates too quickly to respond to changes at the community scale. Rather, patterns of mortality at a larger spatial scale may be important for linking populations to the community. In the language of hierarchy theorists, the extent of our focus should be the community and the grain, a primary unit of demographic change. The appropriate grain for a rhizomatous grass may be the leaf, tiller, plant or larger unit.

To understand plant demography in restored communities, I propose that it is first necessary to detect the proper grain of resolution. In restored grasslands, it is necessary to ask at what scale do rhizomatous grasses respond to differences among restoration treatments. To test this approach I simultaneously monitored the demographics of a rhizomatous grass at four spatial scales. Once an appropriate scale of resolution was detected, demographic data from the rhizomatous grass and a non-clonal bunchgrass were used to explain community dynamics in a restored grassland.

A common restoration activity in Alberta is revegetation of wellsites in native mixed grass prairie (MGP). Restoration could be improved by a stronger fundamental understanding of community dynamics in these systems. The objectives of this study were to 1) determine the appropriate scale for measuring the demography of a rhizomatous grass in response to revegetation treatments and 2) use plant demographic information at an appropriate scale to explain and predict community trends on revegetated MGP wellsites using *Stipa comata* Trin & Rupr. (needle and thread) and *Agropyron smithii* Rybd. (western wheatgrass).

2.0 Methods

2.1 Site description

The study took place in four sites of native prairie in southeastern Alberta, Canada from May 1999 to August 2000 (Figure A-1). Two sites were located 300 m apart, approximately 20 km northwest of Medicine Hat on loam textured Orthic Brown Chernozemic soils. The remaining two sites were 3 km apart, 15 km northeast of Bow Island on sandy loam textured Orthic Brown Chernozemic soils. Natural vegetation in this area is referred to as Dry Mixed Grass Prairie (Strong and Leggat 1992) or MGP (Coupland 1961). Undisturbed prairie at each of the sites was dominated by *Stipa comata* Trin & Rupr. (needle and thread) and *Bouteloua gracilis* (HBK) Lag. (blue grama). Taxonomy follows Moss (1983). The climate is semiarid and the majority of growing season precipitation falls in June. May to September precipitation was slightly below long-term averages in 1996 to 1997 and moderately above long-term averages in 1999 (Table A-1). In 2000 precipitation was approximately half the long-term averages.

Each of the research sites was a drilled and abandoned wellsite located inside a field of native prairie used for grazing. Cattle have grazed each of the fields for over thirty years. In 1999, the range condition of the sites was estimated to vary from good to excellent (Wroe et al. 1988).

Wellsites were established in either fall 1995 or spring 1996 and abandoned in spring 1996. Each of the companies contributing wellsites was asked to reclaim them using normal practices up to the point of being seeded. Each wellsite had the topsoil and subsoil separately removed and stockpiled. After industrial activities were finished, topsoil and subsoil were replaced and a seedbed prepared. Each wellsite and an adjacent strip of undisturbed native prairie was fenced to exclude grazing for the first three years of the study.

2.2 Treatments and experimental design

Three seeded and one non-seeded (Natural Recovery) treatments were compared to the Undisturbed treatment in this study. The seeded treatments consisted of a three species, wheatgrass dominated mix (Current), a five species mix (Simple) and a 21 species mix (Diverse) (Table A-2). Each of these treatments was investigated with and without cattle grazing.

Each of the wellsites was treated as a block. The seeded and Natural Recovery treatments were placed in the same location within each block because of the influence of predominant wind direction on natural revegetation (Table A-2). The position of the Undisturbed treatment varied with each block. Exclosures allowed comparison of Ungrazed and Grazed areas in each revegetation treatment. The experiment was treated as a strip plot design. For this chapter, only Grazed and Ungrazed Natural Recovery, Undisturbed and Simple treatments were investigated.

2.3 Treatment application

Revegetation treatments were implemented in spring 1996 by Hammermeister (2001). Seeding rates (kg/ha) were calculated based on a desired number of pure live seeds (PLS) per unit area (300 PLS/m^2). Because purity and viability data were not available for the forbs, they were estimated as 66%. Chick starter was used as a carrier to ensure even seed distribution and improve seed flow through the drill openers. The total seeding rate was applied with two perpendicular passes of a calibrated Truax native seed drill. *Koeleria macrantha* (Ledeb.) J.A. Schultes f. (june grass) was broadcast with chick starter as a carrier following seeding. Although the seeder was calibrated, the seeding rate varied slightly due to limited accuracy of openers (10%). The species seeded in the simple treatment were *Agropyron smithii* Rydb (western wheatgrass), *Agropyron dasystachyum* (Hook) Scribn (northern wheatgrass), *Stipa comata*, *Koeleria macrantha* and *Bouteloua gracilis*. After seeding a straw crimp was applied to seeded and Natural Recovery treatments by spreading straw bales and mechanically crimping straw into the soil.

During summer 1999 grazing exclosures were built in the center of each site enclosing part of each revegetation treatment. The unfenced portion of each treatment was grazed in June and July 1999. Grazing treatments were applied by moving cattle from the adjoining field into the fenced wellsite where the animals were free to graze any treatments outside the exclosure. Water was provided in troughs placed directly beside the gate of each wellsite. Cattle remained in the wellsite until visually estimated utilization approached 30 to 50% on each treatment. A confounding factor to the experimental design is that two sites were mowed two weeks prior to grazing in 1999. Because of this complication, stocking rates of the Grazing treatment were

reduced for these two sites (Table A-3). In summer 2000 mowed and unmowed sites were visually similar and all blocks were included in the analysis.

2.4 Demographic measurements

Population data can be assembled from field data by following a cohort of individuals through an extended period of time (West et al. 1979, Canfield 1957) or by marking individuals at different stages of the life cycle and following them over a single year (Pyke 1990, Treshow and Harper 1974). This second approach is more suited to a short term study such as this one.

Stipa comata and *Agropyron smithii* were selected for demographic measurements because both species were present in all the treatments investigated and because both are considered important species in undisturbed MGP. Plants were systematically marked in July 1999 along randomly located line transects. Within each grazing/revegetation treatment combination one corner was arbitrarily chosen for the starting point. At two random distances from the corner along one side of the plot transects were established and run perpendicular to the side of the plot. At 2 to 3 m intervals along each transect the nearest individual of each species was permanently marked for a total of 10 marked individuals of each species in each grazing/revegetation treatment combination. In one grazing/revegetation treatment combination, 2 additional individuals were accidentally marked and thus were included in data analysis. When uncommon, individuals of *Stipa comata* or *Agropyron smithii* had to be found by reconnaissance. This occurred in each Natural Recovery treatment and one Undisturbed treatment. Plants were marked by attaching a small label with a unique number to the plant.

For *Stipa comata*, the basal diameter, canopy diameter and number of intact reproductive culms were recorded. Two perpendicular measurements of basal and canopy diameters were made and averaged on each plant. A reproductive culm was counted if the seed head had emerged but all seed had not yet been dropped. In 1999, reproductive culms over 15 were not counted on one of the sites it was not possible to count all culms given the time available to complete data collection .

For *Agropyron smithii* tillers within 3 cm of each other were operationally defined as a plant, while those produced further than 3 cm away were considered a new plant. Although it is recognized that any set of tillers within 3 cm of each other may or may not be genetically identical or physiologically integrated, this operational definition of a plant was based on field observations of the architecture of this species. Additionally, each set of plants within a 9 cm diameter was operationally defined as a "plant unit" (Figure 3-1). For each marked *Agropyron smithii* plant, a single tiller was marked and the number of leaves counted. A leaf was not counted if it had been grazed or senesced half of its length or if it had not yet emerged from the sheath of the previous leaf. Number of tillers in the plant and number of plants in the plant unit to which the marked tiller belonged were counted. Any existing plants within the plant unit were marked with small wires. For each marked plant, the number of intact seed heads was recorded. In June 2000, all marked plants were censused and measurements repeated. Any plants (10%) that could not be found or from which tags became detached were excluded from the data set.

2.5 Data analysis

Two kinds of demographic parameters, state and rate variables, were calculated (Table 3-1). State variables described the structure of a population at a

point in time. For example, the average number of tillers per plant in 1999 is a state variable. Rate variables describe how the structure of a population is changing through time and so are more valuable for indicating real differences of processes between communities and treatments. In this study rate variables were calculated as percent change, Lambda (λ). For example, if the average number of tillers per plant was 2 in 1999 and 3.5 in 2000, the lambda rate of change would be 1.75 (3.5/2).

Data from all plants within a plot were pooled to calculate a number of parameters for each plot. Data were analyzed as a strip plot design using analysis of variance (ANOVA) (Appendix III). Treatment means were compared using the least significant difference (LSD) at alpha equal to 0.05. SAS software (SAS Institute Inc. 2000) was used to analyze the data using Proc GLM. Proc GLM uses the wrong error terms for split designs and so correct F-tests were obtained by use of a random statement and specifying the error term in mean comparison tests. In cases where interactions had to be investigated, Proc Mixed was used to investigate comparison of least squared means.

Given the small sample size, tests were not made for normality or homogeneity of variance. ANOVA is generally regarded as robust to violations of both of these assumptions (Day and Quinn 1989). The third assumption of ANOVA, independence of samples, is also violated in this study by the lack of randomization. The fixed location of treatments in this study increases the risk of a Type I error. However, many precedents exist to justify this analysis in field studies and choice of analysis should reflect the judgment and objectives of the researcher rather than inflexible statistical rules of thumb (Stewart-Oaten 1995).

3.0 Results and Discussion

3.1 General results

A total of 215 *Stipa comata* plants of 240 originally marked were used to calculate demographic measurements. Only 213 plants were used to calculate number of reproductive culms because of missing data from the second census. *Stipa comata* plants in the Natural Recovery treatment had a larger basal diameter than other revegetation treatments in 2000 (Table 3-3). Plants in the Undisturbed treatment had lower survival than in other revegetation treatments. Grazing treatments did not have a significant effect on any of the variables.

A total of 218 plants of the original 242 marked were used in the calculation of demographic parameters for *Agropyron smithii*. The exception was at the leaf scale where only 202 marked tillers were used because of missing data from the second census.

Significant effects ($p = 0.002$) were found for grazing treatment on leaves per tiller in 1999. In 1999, leaves per tiller were higher in the Ungrazed treatment (3.4) than the Grazed treatment (2.4) (Standard error = 0.12). Revegetation by grazing treatment interactions for the lambda of leaves were significant, but were not investigated further. The Natural Recovery treatment had significantly more tillers per plant than the Undisturbed treatments (Table 3.3).

In 2000, the Undisturbed treatment had significantly fewer plants per plant unit than the Natural Recovery or Simple treatments. The lambda of number of plants per unit and number of plant recruits was highest in the Natural Recovery, then Simple and finally Undisturbed treatment. Survival of plant units in the Undisturbed

treatment was significantly lower than in the Simple and Natural Recovery treatments (Table 3-3).

3.2 Adequacy of the data set

The type of experimental design used to evaluate population dynamics of herbaceous plants varies widely. Long-term studies typically use permanently marked quadrats and a pantograph-mapping table to chart the position of all individual plants. Repeated over a long time entire cohorts can be followed from birth to death and associated life tables constructed. These studies often have a large number of marked individuals (e.g. 18,000, Mack and Pyke 1983) and complex statistical designs to account for overlapping cohorts (e.g. Bartlett and Noble 1985). Other studies reconstruct life tables from excavated root systems (Callaghan 1976).

A contrasting method to measuring demographic patterns is to mark plants and follow them over a short period of time to determine patterns of death and reproduction (e.g. Bernard 1976). These studies typically have a smaller number of marked individuals comparable to this study. The experimental design in this study is most comparable to that of Zhang and Romo (1995) who marked all tillers of *Agropyron dasystachyum* inside a single 100 cm^2 quadrat inside each plot. Rather than mark all tillers within a single permanent location, we marked a single tiller at 10 locations in each subplot and then evaluated the number of plants in a 64 cm^2 area. The number of tillers marked in this study is therefore similar to that used by Zhang and Romo (1995) as well as Harrison and Romo (1994) (5 per plot). The ability of this study to detect statistically significant treatment differences also attests to the adequacy of the sample size.

This study did not separate death and birth rates between different components of the population of each species although it is recognized that these rates are usually age specific (Bazzaz and Harper 1977). Neighborhood effects, density dependence and size hierarchies were explicitly ignored in the treatment of the data because the objective was to detect processes at the community scale rather than at the neighborhood scale.

3.3 Determining the scale of demographic processes in *Agropyron smithii*

Plant population demographics occur across a hierarchy of scales (Harper and White 1972). Most studies of plant demography however, have focused only on one scale (e.g. Reekie and Redmann 1991, Zhang and Romo 1995), or the genet and ramet (tiller) scales together (Kays and Harper 1974, Bernard 1990). Few studies have looked at more than two scales simultaneously (e.g. Hartnett and Bazzaz 1985a). This study looked at four levels of demographics simultaneously; leaf, tiller, plant and plant unit. While complete information was not collected across all of these scales (e.g. no information was gathered on individual leaf turnover rates), we have the opportunity to observe population characteristics and processes across a range of scales. These processes were observed in three different revegetation treatments among which we would expect very different population patterns to be occurring. We can then test for the grain of demographic processes at the extent of plant community. In other words we are trying to detect the spatial scale at which important population processes occur in a rhizomatous grass in response to plant community differences or grazing treatments.

To detect processes at various scales we must focus on rate variables. State variables may tell us if there are structural differences between populations but will

not tell us if there are important temporal trends in these characteristics. In response to plant community treatments, significant rate variables occur only at the plant and plant unit scales. The highly significant (lambda) rates of change for the number of plants per module provides strong evidence that important processes are occurring at this scale. The lack of significance at the tiller scale shows that the operational definition of a plant comprising tillers 3 cm apart reflects a real pattern. The processes driving this system are not apparent at the tiller scale. This is similar to Bernard's (1990) suggestion that the important unit of demography in some *Carex* species may be clumps of tillers. Survival was also significant at the plant unit scale, however, we measured only a single demographic parameter at this scale. We did not measure rates of change of the number of modules in the next scale of hierarchy or patterns of recruitment. Changes in the abundance of *Agropyron smithii* can be explained by observing at the scale of plant or possibly plant unit.

Among researchers who have investigated plant population processes at more than one scale, results have generally supported the importance of detecting the appropriate grain. Hartnett and Bazzaz (1985b) found that population processes in *Solidago canadensis* L. (Canada goldenrod) occurred at the entire clone (genet) scale in a successional old field in eastern North America. Abdul-Faith and Bazzaz (1980) found density dependent effects at the plant and leaf scale in *Ambrosia trifida* L. (great ragweed). Butler and Briske (1988) examined both tiller and plant scale population processes in *Schizachyrium scoparium* (Michx.) Nees (little bluestem) and found differential responses to different processes in the community.

Most studies of rhizomatous grasses examine only the tiller scale (e.g. Cullan et al. 1999). The danger of this approach is lack of similarity of response to plant and

community spatial scales. For example, Zhang and Romo (1995) investigated tiller survival in *Agropyron dasystachyum* (a species similar in architecture and ecology to *Agropyron smithii*) and observed significant effects of defoliation treatments on tiller survival and recruitment. However it is unclear if their results are applicable to the larger community as they sampled only a 100 cm^2 quadrat in each plot. The extent of their investigation is technically only 10 by 10 cm and the grain a single tiller. If they had randomly marked tillers across the entire community, a different response may have been observed. Alternately, they may have had to look at the plant scale to see a significant response. The difficulty in interpreting these results illustrates the importance of detecting the biological scale at which plant responses occur.

3.4 Response of demographic patterns to grazing treatments

Investigators in arid and semiarid rangelands have found strong relationships between grazing and plant demographic parameters. Cattle grazing impacted demographic patterns of species in salt desert shrublands (Chambers and Norton 1993), Australian rangeland (Williams 1970), Arizona semi-desert (Canfield 1957) and sagebrush steppe (West et al. 1979). Grazing generally impacted negatively on the demography of preferred grazing species and positively on the demography of less preferred species. Both the species investigated in this study are preferred grazing species by cattle in northern MGP (Wroe et al. 1989). Grazing may also affect populations by changing the average size of individuals. Butler and Briske (1988) found that grazing reduced the average size of plants by breaking up clones. Peterson (1962) found that defoliation reduced the average size of *Stipa comata.*

In this study, the only significant effect of grazing treatment was the number of leaves per tiller in 1999 and the lambda of leaves per tiller in the Simple and

Natural Recovery treatments. This result makes intuitive sense in that grazing should reduce the number of leaves per tiller if cattle are selectively removing plant parts. No other studies reporting the impact of cattle grazing on the number of leaves per tiller were found. Water stress significantly impacted leaf demography in *Agropyron dasystachyum* (Reekie and Redmann 1991). Defoliation is also documented to impact the average weight of tillers and leaves (Osterheld and McNaughton 1988, Li and Redmann 1992). Laurenroth et al. (1985) found that defoliation increased the number of tillers per plant in *Agropyron smithii*. Since grazing treatments were applied for only a single year in this study, it is not surprising that little population response was seen to grazing.

3.5 Response of demographic patterns to revegetation treatments

Revegetation treatments had a larger impact on demographic parameters than did grazing treatments. Two state variables showed significant differences but did not show a corresponding difference in rate variables (*Stipa* Basal 2000, *Agropyron* Tillers 1999) possibly reflecting differences in abundance or architecture rather than demographic differences. For *Agropyron smithii*, we can conclude that the Natural Recovery treatment has a higher tiller density than the Undisturbed treatment. However we cannot assume that any processes of birth or death are different. For *Stipa comata*, we can say that plants in the Natural Recovery treatment are larger than in the other treatments.

Although revegetation treatment effect was only near significant (p=0.08) number of seed heads per plant in 1999 was higher in the Natural Recovery (6.2) than the Undisturbed treatment (1.7) (standard error = 1.6). Pyke (1990) investigated demographic patterns of bunchgrasses and concluded that the production of more

seed culms by *A. desertorum* (Fisch. ex Link) Shultz. (crested wheatgrass) would likely lead to expansion of this species. In a similar fashion this study may have evidence that the production of more seed culms in the Natural Recovery treatment will lead to faster expansion of this species.

Rate variables allow us to demonstrate some important differences in demographic processes between revegetation treatments. Survival of *Stipa comata* plants was lower in the Undisturbed plant community. Overall, survival of *Stipa comata* was surprisingly high. Scheiner (1988) found that in the perennial bunchgrass, *Danthonia spicata*, that half of the population died every 2 to 3 years. For *Agropyron smithii*, the number of plants per plant unit was increasing faster in the Natural Recovery, then Simple and finally Undisturbed treatments (λ plant). The mechanism of this increase was not the survival of plants but the recruitment of new plants into the module. That the number of plants per module was not significantly different in 1999 but was in 2000 further indicates that this difference is not a matter of differences in abundance or architecture between treatments but actually a difference in population process. Further evidence that demographic differences exist between the communities is provided by the significantly lower survival rate of modules in the Undisturbed treatment.

3.6 Demographic patterns and community dynamics

If we consider the Natural Recovery treatment an early seral community, the Undisturbed treatment a late seral community and the Simple treatment intermediate we can generalize about the demographic patterns observed. Survival appears to be lower in the late seral environments and reproduction seems to be higher in the early seral communities. These patterns are logical. In early successional environments,

generally resources are more abundant and consequently populations would be rapidly expanding to take advantage of the available resources. In later successional environments (i.e. Undisturbed treatment) plant populations may have come to an equilibrium in abundance and so population turnover rates are more stable. Apparently even in the densely seeded Simple treatment, resources are abundant enough that population growth rates are high. Scheiner (1988) compared the population dynamics of *Danthonia spicata* (L.) Beauv. (poverty oat grass) in forest stands of different successional stages and found that local variations in environment were more important to demography than the seral stage of the plant community. Evidence from this study supports the argument that population processes are constrained by the plant community.

An alternative perspective to explain the differences between successional environments may be that different ecotypes may be present in different environments. For example Harper (1977) presented data demonstrating that different ecotypes of a *Solidago* spp. were present in different stages of old field succession. Ecotypes that are adapted to rapid vegetative reproduction may be invading the Natural Recovery treatment. The cultivar of *Agropyron smithii* seeded in the Simple treatment has been bred for aggressive agronomic characteristics (Smoliak and Johnston 1983). The *Agropyron smithii* present in the Undisturbed treatment may be adapted to low growth rates as a survival mechanism.

3.7 Predicting future population changes

The ultimate aim of understanding plant population processes at the community scale is to be able predict future plant population trends. This study measured demographic information and while the information has been sufficient to

characterize population processes across scales and seral communities, it is unclear as to how well this information will predict future changes in various treatments. The population data indicate that likely *Stipa comata* and *Agropyron smithii* should be increasing most quickly in the Natural Recovery treatment, either increasing or stable in the Simple treatment and stable in the Undisturbed treatment. Cover estimates of both these species have similar trends (Chapter V). *Agropyron smithii* and *Stipa comata* are both increasing in the Natural Recovery while remaining stable in the Undisturbed and Simple treatments.

This study measured demography over a single year. However, short-term assessments of population parameters may have value for predicting future vegetation trends. For example, range managers sometimes use number of young seedlings of desired species to predict future rangeland composition (CRC 1995). A critical step is choosing the appropriate grain for measuring population processes.

4.0 Conclusions

Population processes in the rhizomatous grass *Agropyron smithii* were different among revegetation treatments at the plant and plant unit scales. Grazing treatments did not impact the demography of either *Stipa comata* or *Agropyron smithii* after a single year. In the Natural Recovery treatment (early seral) both species were generally reproducing faster and surviving better than in the Undisturbed or Simple (seeded) treatments. These species will likely continue to increase in the Natural Recovery treatment. Short term assessments of population parameters may be useful for predicting community trends if the appropriate "grain" for the plant community "extent" can be identified.

Table 3-1: Demographic parameters for two prairie grasses on revegetated wellsites in mixed grass prairie.

Variable	Variable Type	Formula	Description
Stipa comata			
Basal 2000	state	NA[1]	Mean basal radius for all marked plants in plot in 2000
Basal 1999	state	NA	Mean basal radius for all marked plants in plot in 1999
λ Basal	rate	Basal 2000 ÷ Basal 1999	Absolute change in basal radius
Canopy 2000	state	NA	Mean canopy radius for all marked plants in subplot in 2000
Canopy 1999	state	NA	Mean canopy radius for all marked plants in subplot in 1999
Seed culms 2000	state	NA	Mean number of intact reproductive culms for all marked plants in subplot
Seed culms 1999	state	NA	Mean number of intact reproductive culms for all marked plants in subplot
λ Seed culms	rate	Seed culms 2000 ÷ Seed culms 1999	Absolute rate of change in number of seed culms
Survival	rate	#surviving plants ÷ #marked plants	Percent survival of *Stipa* plants
Agropyron smithii			
Leaves 2000	state	NA	Mean number of leaves of all marked tillers in subplot in 2000
Leaves 1999	state	NA	Mean number of leaves all marked tillers in subplot in 1999
λ Leaves	rate	Leaves 2000 ÷ Leaves 1999	Relative rate of change
Tillers 2000	state	NA	Mean number of tillers of all marked plants in subplot in 2000
Tillers 1999	state	NA	Mean number of tillers of all marked plants in subplot in 1999
λ Tillers	rate	Tillers 2000 ÷ Tillers 1999	Relative rate of change of number of tillers per plant
Tiller survival	rate	#surviving plants ÷ # marked plants	Percent survival of *Agropyron* tillers
Plants 2000	state	NA	Mean number of plants per marked plant unit in subplots in 2000
Plants 1999	state	NA	Mean number of plants per marked plant unit in subplots in 1999
λ Plants	rate	Plants 2000 ÷ Plants 1999	Relative rate of change of number of plants per plant unit
Plant survival	rate	# surviving plants÷ #plants marked	Percent survival of marked *Agropyro*n plants
Pooled survival	rate	# surviving plants ÷ #plants marked	Percent survival of all plants in plant unit
Plant recruitment	rate	NA	Number of new plants established in module in 2000
Unit survival	rate	# surviving units ÷ # marked units	Percent survival of *Agropyron* units

[1]No calculation necessary, direct measurement

Table 3-2: P values for factors in strip plot analysis of variance for demographic parameters for two grasses on revegetated wellsites in mixed grass prairie.

Scale	Variable	Block	Trt[2]	Graze	Trt*Graze	Block*Graze	Block*Trt
Stipa comata							
Plant	Basal 2000	0.16	**0.007**[1]	0.49	0.57	0.29	0.20
	Basal 1999	0.37	0.25	0.99	0.55	0.12	0.45
	λ Basal	0.66	0.40	0.57	0.94	0.15	0.39
	Canopy 2000	0.27	**0.04**	0.89	0.70	**0.05**	0.19
	Canopy 1999	0.22	0.13*	0.97*	0.47	**0.03**	0.56
	λ Canopy	0.79	0.31	0.40	0.77	0.40	0.39
	Seedheads2000	0.47	0.37*	0.33*	0.56*	**0.001**	0.52
	Seedheads1999	0.16	0.08	0.52	0.67	0.11	0.24
	λ Seed culms	0.49	0.42	0.43	0.37	0.06	0.42
	Survival	0.82	0.06*	0.81*	0.95*	0.92	0.45
Agropyron smithii							
Leaves	Leaves 2000	0.85	0.79	0.32	0.08	0.06	0.11
	Leaves 1999	**0.0001**	0.63	**0.0001***	**0.01***	0.94	0.84
	λ Leaves	0.54	0.63*	0.11	**0.04**	0.06	0.08
Tiller	Tillers 2000	0.55	0.13	0.45	0.06	**0.001**	0.15
	Tillers 1999	0.54	**0.03**	0.89	0.30	**0.003**	**0.007**
	λTiller	0.70	0.65	0.95	0.77	**0.003**	0.08
	Survival	0.96	0.76	0.57	0.07	0.06	**0.05**
Plant	Plants 2000	0.67	**0.005**	0.06	0.42	0.07	0.11
	Plants 1999	0.57	0.59*	0.94	0.32	0.19	0.76
	λ Plants	0.45	**0.0007**	0.40	0.51	0.43	0.41
	Recruits	0.87	**0.003**	0.90	0.56	0.11	0.07
	Survival	0.88	0.15	0.91	0.55	0.38	0.06
	Pooled survival	0.96	0.41	0.95	0.55	0.16	0.06
Module	Survival	0.87	**0.001***	0.53	0.18	0.80	0.28

[1]Bolded values are significant at alpha equal to 0.05
[2]Revegetation treatments
*F-tests rerun in proc mixed

Table 3-3: Revegetation treatment means for select population variables on revegetated wellsites in mixed grass prairie, 1999 to 2000.

Variable	Standard Error	Natural Recovery	Simple	Undisturbed
Stipa comata				
Basal 2000 (cm)	0.37	3.9a[1]	3.0b	2.1b
Survival	0.02	1.0a	1.0a	0.95b
Agropyron smithii				
Tillers 1999	0.42	2.8a	1.9ab	1.3b
Plants 2000	0.35	2.7a	1.9a	0.7b
λ Plants	0.06	2.1a	1.5b	0.8c
Recruits	0.31	2.2a	1.2b	0.4c
Module survival	0.03	0.99a	0.98a	0.83b

[1]Means with different letters are significantly different

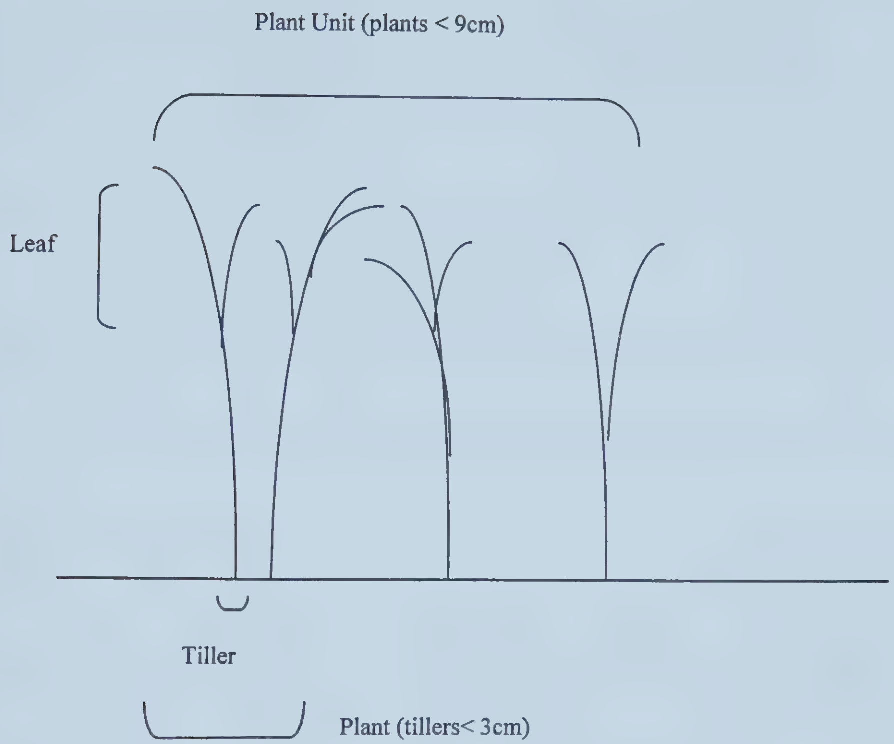

Figure 3-1: Operational hierarchy of parts for *Agropyron smithii.*

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Chapter IV Community Trends in Mixed Grass Prairie Wellsite Revegetation

1.0 Introduction

Restoration activities often focus on establishing a particular plant community. The desired plant community may be the predisturbance state, a cultural landscape, or an indigenous/socially desirable assemblage of plants. Traditionally, the establishment of these plant communities was seen as a purely technical (Bradshaw 1987), agronomic or horticultural (Allen and Hoekstra 1987) problem. More recently however, restoration of plant communities has been conducted in the framework of plant succession. Restoration is seen as simply manipulating or directing the inherent tendency of plant communities to change over time (Call and Roundy 1991). Weed control is seen as a method of accelerating the process of species replacement. Seeding accelerates the process of migration to a disturbed site. In these respects, restoration is managed succession.

To achieve restoration goals and develop effective management techniques a strong understanding of successional processes is required. Restoration ecology must be fundamentally grounded in basic research (Bradshaw 1987). A fundamental understanding of processes governing succession should allow restoration practitioners to develop optimal restoration techniques (Call and Roundy 1991).

Two major approaches to understanding community change can be discerned in the literature. Most of the literature focuses on determining mechanisms for the gradual, sometimes predictable, change of communities from one stage to another. Individual researchers emphasize site specific events (Gleason 1926), species

availability (Egler 1954), resource ratios (Tilman 1988), life history characteristics (Grime 1977), species interactions (Connell and Slatyer 1977) or a range of complex factors (Picket et al. 1987). Despite often fierce debate among these researchers, they are all united by a paradigm that sees vegetation change as a continuum of stages from simple ruderal communities to more complex communities dominated by perennials (Clementsian succession). What remains contentious in this school of thought is what autogenic mechanisms (e.g. population characteristics, competition, disturbance) determine the commonly observed patterns of community change.

The second approach in the literature recognizes that external forces are more important to understanding vegetation changes. Rather than form a continuum of directional stages, vegetation is seen to remain in one of several states or domains. External energy or disturbance is required to push the community past a particular threshold, whereby the community rests in a new state (Friedel 1991). The transition to the new state may or may not be reversible. This approach has been variously called state and transition models (Westoby et al. 1989) or ball and cup models (Laycock 1991). While the applications of this model lack strong theory they have been useful on a system specific basis for explaining vegetation change (Rodriguez-Iglesias and Kothman 1997). These applications have proven particularly useful in arid and semiarid rangelands (Friedel 1991). For example, Laycock (1991) used a state and transition model to describe vegetation change in Great Basin sagebrush range that did not fit the traditional Clementsian model.

Mixed grass prairie (MGP) is a semiarid grassland where a common restoration activity is the revegetation of oil wellsites. Revegetation efforts typically include seeding native wheatgrass cultivars in an attempt to reestablish

predisturbance communities (Gerling et al. 1996). Alternative proposed revegetation practices include not seeding (Natural Recovery), including more species in the seed mixes and introducing grazing early in the revegetation process as a management strategy. The necessity of managing restoration projects has been widely recognized (Wark et al 1996) and cattle grazing presents an obvious management option for MGP wellsites. The long-term consequences of these practices on community composition of these sites are unknown however. Either traditional successional theory or state and transition models may help explain and predict community trends on revegetated MGP wellsites.

The direction of succession during drought in the MGP is well documented and more or less follows conventional successional theory. Generally, *Bouteloua gracilis* (HBK) Lag (blue grama grass) increases in dry conditions while *Agropyron* species and *Stipa* species decrease (Coupland 1961). Similarly, under heavy grazing pressure *Bouteloua gracilis* increases and *Agropyron* and *Stipa* species decrease (Dormaar et al. 1994, Smoliak 1974). The mechanisms of these predictable changes are not well understood however. For example, Smoliak et al. (1972) attempted to explain MGP successional change under grazing in terms of soil property changes but could not reach any strong conclusions. In some cases MGP succession does not appear to follow traditional theory but rather the kind of behavior predicted by state and transition models. For example, Dormaar et al. (1994) found that abandoned cultivation rather than returning to the typical MGP composition, became a community completely dominated by *Stipa comata* Trin. & Rupr. (needle and thread). In a similar fashion, native MGP that had been grazed heavily for a long

period of time became dominated by *Bouteloua gracilis* but when protected from grazing, would not return to its previous condition (Dormaar et al 1994).

To investigate community development of MGP after industrial disturbance, 3 seeded and a Natural Recovery (unseeded) treatment were applied to 7 wellsites on native prairie in southeastern Alberta. Community composition was assessed for 5 consecutive years and a grazing treatment was applied to four of the sites in the fourth and fifth year. The objectives of this research were to 1) test whether traditional successional and state and transition models could explain community trends on revegetated wellsites in MGP and 2) use traditional successional and state and transition models to predict future community trends on these wellsites.

2.0 Methods

2.1 Site description

The study took place in seven native prairie sites in southeastern Alberta, Canada from May 1999 to August 2000 (Figure A-1). Two sites were located 300 m apart, approximately 20 km northwest of Medicine Hat on loam textured Orthic Brown Chernozemic soils. Two sites were 3 km apart, 15 km northeast of Bow Island on sandy loam textured Orthic Brown Chernozemic soils. Three sites on Solonetzic sites were located within 50 km of Brooks. Natural vegetation in this area is Dry Mixed Grass Prairie (Strong and Leggat 1992) or MGP (Coupland 1961). Undisturbed prairie at each of the sites was dominated by *Stipa comata* Trin & Rupr. (needle and thread) and *Bouteloua gracilis* (HBK) Lag. (blue grama). *Agropyron dasystachyum* (Hook) Scubn (northern wheatgrass) is also a dominant grass on Solonetzic soils. Taxonomy follows Moss (1983). The climate is semiarid and the

majority of growing season precipitation falls in June. May to September precipitation was slightly below the long term average from 1996 to 1997 and moderately above long-term averages in 1999 (Table A-1). In 2000 precipitation was approximately half the long-term averages.

Each of the research sites was a drilled and abandoned wellsite located inside a field of grazed native prairie. Cattle have grazed each of the fields for over thirty years. In 1999, the range condition of the sites was estimated to vary from good to excellent (Wroe et al. 1988).

Wellsites were established in either fall 1995 or spring 1996 and abandoned in spring 1996. Companies contributing wellsites were asked to reclaim them using normal practices up to the point of being seeded. Each wellsite had the topsoil and subsoil separately removed and stockpiled. After industrial activities were finished, topsoil and subsoil were replaced and a seedbed prepared. Each wellsite and an adjacent strip of undisturbed native prairie was fenced to exclude grazing for the first three years of the study.

2.2 Treatments and experimental design

Three seeded and one non-seeded (Natural Recovery) treatments were compared to the Undisturbed treatment. The seeded treatments consisted of a three species, wheatgrass dominated mix (Current), a five species mix (Simple) and a 21 species mix (Diverse) (Table A-2). On Chernozemic sites, each of these treatments was further investigated with and without cattle grazing.

Each wellsite was treated as a block. The 3 seeded and Natural Recovery treatments were placed in the same location within each block because of the

influence of predominant wind direction on natural revegetation (Table A-2). The position of the Undisturbed treatment varied with each block. Exclosures allowed comparison of Ungrazed and Grazed areas within each revegetation treatment.

2.3 Treatment application

Revegetation treatments were implemented in spring 1996 by Hammermeister (2001). Seeding rates (kg/ha) were calculated based on a desired number of pure live seeds (PLS) per unit area (300 PLS/m^2). Because purity and viability data were not available for the forbs, they were estimated as 66%. Chick starter was used as a carrier to ensure even seed distribution and improve seed flow. The total seeding rate was applied with two perpendicular passes of a calibrated Truax native seed drill. *Koeleria macrantha* (Ledeb.) J.A. Schultes f. (june grass) was broadcast with chick starter as a carrier. Although the seeder was calibrated, the seeding rate varied slightly due to limited accuracy of openers (10%). After seeding a straw crimp was applied to seeded and Natural Recovery treatments by spreading straw bales and mechanically crimping straw into the soil.

During summer 1999 grazing exclosures were built in the center of each Chernozemic site enclosing part of each revegetation treatment. The unfenced portion of each treatment was grazed in June and July 1999. Grazing treatments were applied by moving cattle from the adjoining field into the fenced wellsite where the animals were free to graze any of the treatments outside the exclosure. Water was provided in troughs placed directly beside the gate of each wellsite. Cattle remained in the wellsite until visually estimated utilization approached 30 to 50% on each treatment. A confounding factor to the experimental design is that two of the sites

were mowed two weeks prior to the application of the grazing treatment in 1999. Because of this complication, stocking rates of the Grazing treatment were reduced for these two sites (Table A-3). In summer 2000 mowed and unmowed sites were visually similar and all blocks were included in the analysis. Summer 1999 represented a period of above average precipitation while 2000 was a year of below average precipitation so stocking rates were lower in 2000. No grazing treatments were applied to the Solonetzic sites.

2.4 Data collection

In 1997 and 1998 plant species composition data were collected from thirty 0.1 m^2 quadrats placed at even intervals along three parallel transects in each treatment (Hammermeister 2001). Species canopy cover was estimated to the nearest percent in each quadrat. In 1999, because grazing exclosures were built on the Chernozemic sites, the vegetation sampling scheme was modified. In each plot one corner was arbitrarily chosen for the starting point. Transects were run at three random distances from the corner along one side of the plot. Each of the three transects ran perpendicular to the side of the plot. At three (Chernozemic sites) or six (Solonetzic sites) m intervals along each transect four 0.1 m^2 quadrats were placed for a total of twelve quadrats in each plot (treatment combination). Species area curves performed on 1997 data indicated that 12 quadrats was sufficient to describe communities. Quadrats were placed further apart on Solonetzic sites because experimental units were larger than on Chernozemic sites. Species cover was estimated along an eleven point cover scale (1 = present to 5%, 2 = 6 to 10%, 3= 11

to 20% 4 = 21 to 30%, 5= 31 to 40%, 6= 41 to 50%, 7= 51 to 60%, 8= 61 to 70%, 9= 71 to 80%, 10= 81 to 90%, 11= 91-100%) in each quadrat.

2.5 Data analyses

Species composition data from 1997 and 1998 were converted into the 11 point scale used for cover estimates in 1999 and 2000. Quadrats were then averaged within each plot to give composition data for each plot in each year. Reciprocal averaging was used to generate an ordination of Solonetzic sites and Chernozemic sites separately using PC-Ord software (McCune and Mefford 1997).

Cover scores of seven dominant species were analysed using analysis of variance (ANOVA). Solonetzic sites were analyzed as a randomized complete block design. Chernozemic sites were analyzed as a randomized complete block design for 1997 and 1998 and as a strip plot for 1999 and 2000 (Appendix III). Treatment means were compared using the least significant difference (LSD) at alpha equal to 0.05. SAS software was used to analyze the data using Proc GLM (SAS Institute Inc. 2000). Proc GLM uses the wrong error terms for split designs so correct F-tests were obtained by use of a random statement and specifying the error term in mean comparison tests. Inspection of mean sums of squares revealed that in some cases the residual error was larger than the block by treatment interactions (error terms A and B). To find correct error terms Proc Mixed was used to recalculate F-tests and conduct mean comparison tests.

Given the small sample size, tests were not made for normality or homogeneity of variance. ANOVA is generally regarded as robust to violations of both of these assumptions (Day and Quinn 1989). The third assumption of ANOVA,

independence of samples, is also violated in this study by the lack of randomization. The fixed location of treatments in this study increases the risk of a Type I error. However many precedents exist to justify this sort of analysis in field studies and choice of analysis should reflect the judgement and objectives of the researcher rather than inflexible statistical rules of thumb (Stewart-Oaten 1995).

3.0 Results and Discussion

3.1 General results

Reciprocal averaging of Chernozemic sites provided good separation of some treatments. Undisturbed treatments were separated from all other treatments along Axis 1 (Figure 4-1). Natural Recovery treatments were separated from seeded treatments along Axis 2. Axis 3 did not provide any additional treatment separation so results are not presented. Visual inspection of the ordination plot does not reveal any apparent arching effect that is an associated artifact of reciprocal averaging (Gauch 1982). No clear separation occurred between the seeded Diverse, Simple and Current treatments.

To investigate individual successional trends within a treatment, each site was plotted for years 1997 through 2000 and connected. In 1999, since grazing treatments were applied the vectors branch into Grazed and Ungrazed segments. Successional vectors of Diverse, Simple and Current treatments all begin close to the origin or in the top left hand quadrant and converge in the bottom left quadrant by 2000 (Figures 4-2 to 4-5). Grazing did not appear to alter the successional trends of the seeded treatments. Successional vectors of the Natural Recovery treatments all appear to start in the same area as the seeded treatments and initially head towards

the Undisturbed treatments (Figure 4-6). In 2000, the Grazed Natural Recovery treatments either continue towards the Undisturbed treatment or head back towards the upper left quadrant. In 2000, the Ungrazed Natural Recovery treatments suddenly head toward the seeded treatments in the lower left quadrant. Each of these treatments had a small amount of *Agropyron dasystachyum* that had invaded from adjacent seeded treatments. Succession towards the seeded treatments is therefore an artifact of the experiment rather than a true successional trajectory. Alternatively, the extreme drought of 2000 may have caused the regression of the treatments.

Reciprocal averaging of Solonetzic sites provided good separation of some treatments. Undisturbed treatments were again separated from all other treatments along Axis 1 (Figure 4-7). Some separation of Natural Recovery treatments from seeded treatments occurred along Axis 2. Axis 3 did not provide any additional treatment separation and so is not presented. No arching effect was evident as an artifact of the ordination (Gauch 1982). No clear separation occurred between the Simple, Diverse and Current treatments.

Successional vectors of the Diverse, Simple, Current and Natural Recovery treatments on Solonetzic soils start in the upper left quadrant for the PCP South and North sites and in the lower left for the PCP West site. By 2000, however, all Diverse, Simple and Current vectors converge in the lower left quadrant (Figures 4-8 to 4-10). All Natural Recovery vectors appear to head directionally for the Undisturbed treatments.

On both Chernozemic and Solonetzic sites, individual species trends were similar (Tables 4-1 and 4-2). *Agropyron dasystachyum* and *Agropyron smithii* were

the dominant grasses in seeded treatments, establishing a relatively constant abundance by 2000. *Agropyron trachycaulum* (Link) Malte (slender wheatgrass) while abundant in the first three years in the seeded treatments, appears to decline by 2000. None of these three grasses were as abundant in the Natural Recovery or Undisturbed treatments as in the seeded treatments.

Bouteloua gracilis, while very abundant in the Undisturbed treatment, is only a minor component of the plant community in the seeded and Natural Recovery treatments. The abundance of this grass appears to have stabilized in the Simple and Diverse treatments but is declining in the Natural Recovery and Current treatments. However the rapid decline of these grasses may be due to the extreme drought of 2000 rather than any actual trend. *Stipa comata* is a major component of the Undisturbed treatment but a relatively small part of the revegetated plant community. In the seeded treatments this grass appears to be declining in abundance while increasing in the Natural Recovery treatment. However the rapid decline of these grasses may again be due to the extreme drought of 2000 rather than any actual trend. The weedy annuals, *Salsola kali* L. (russian thistle) and *Kochia scoparia* (L.) Schrad. (kochia weed), while abundant in early years on the revegetation treatments appear to have declined in abundance in 1999 and 2000.

Grazing treatments had a significant effect on species only for *Agropyron dasystachyum* in 2000 ($p = 0.02$). However treatment by grazing interactions were also significant for *Agropyron dasystachyum* in 2000 ($p = 0.02$). Grazing significantly reduced cover in the seeded treatments (Simple grazed = 2.6, ungrazed = 3.7, Diverse grazed = 1.6, ungrazed = 2.1, Current grazed = 2.1, ungrazed = 3.4) but not in the Natural Recovery or Undisturbed (standard error = 0.30). Significant

grazing by revegetation treatment interactions were also found for *Stipa comata* in 2000. Least squares mean tests of revegetation treatment by grazing interactions reflected results of mean comparison tests in Table 4-1 so results are not presented.

Significant treatment by block interactions were found for *Agropyron dasystachyum*, *Agropyron trachycaulum*, *Bouteloua* gracilis, *Kochia scoparia* and *Salsola kali* in 1999 and for *Agropyron dasystachyum*, *Kochia scoparia*, *Salsola kali* and *Stipa comata* in 2000. Because the primary objective of analysis was to reflect trends through time only the results of mean comparison tests are presented (Tables 4-1 and 4-2). The block by treatment interactions were not further investigated.

3.2 States and transitions in mixed grass prairie revegetation

The characteristic behavior of state and transition models include identifiable states or domains of attraction in which communities occur and transitions between these states that are sudden, often irreversible and caused by an identifiable external agent (Rodriguez Iglesias and Kothmann 1997). Friedel (1991) argued that multivariate analysis could be used to identify stable states in range ecosystems. Multivariate analysis of revegetation treatments in this study identified several distinct states or domains characteristic of state and transition models. The Undisturbed native prairie was the first of these states. All of the Undisturbed treatments were tightly clustered distinct from any of the revegetation treatments. A second clear domain of attraction was a wheatgrass dominated community composed of the three seeding treatments. The successional trajectories of the seeded treatments provide further evidence for the existence of this state.

States are often described as domains of attraction or as a cup and ball model. That is, ecosystems are attracted to stable low energy states (Laycock 1991). That all the seeded treatments appear to converge by 2000 lends support to the existence of a wheatgrass dominated domain of attraction. Finally, the Natural Recovery treatments represent a final if weaker identifiable state in the ordination analysis. On Solonetzic sites, Natural Recovery treatments, though not tightly clustered, proceeded parallel with each other towards the Undisturbed treatment. On Chernozemic sites, Natural Recovery treatments either moved within the top left quadrant or headed towards the Undisturbed treatments.

The second element of state and transition models is the actual transitions themselves. In this study there are clearly identifiable mechanisms of transition between the states identified by multivariate analysis. The first transition was construction of the wellsite which changed Undisturbed native prairie to a disturbed state (Figure 4 -11). The wellsite disturbance was transferred to a wheatgrass dominated state by seeding wheatgrass cultivars. Inclusion of other species in the seed mix did not move the community out of the domain of attraction of the wheatgrass dominated state. In the absence of seeding (Natural Recovery), the disturbed wellsite moved to a community dominated by weedy dicots and mid-seral grasses.

Other possible transitions are suggested by the data but it is unclear whether they will occur. Eventually the Natural Recovery treatment may progress from being dominated by weedy dicots and midseral grasses and become Undisturbed native prairie. This transition reflects traditional successional theory and is discussed below. A second transition not suggested by the ordination results but by

the individual species data is that grazing may transfer a wheatgrass dominated community to native prairie. Individual species data demonstrated that grazing had significantly reduced the cover of *Agropyron dasystachyum,* the most dominant species in the seeded treatments. Over a long period of grazing, seeded treatments may become compositionally similar to the Undisturbed treatment.

3.3 Applying traditional successional models to prairie revegetation

Traditional linear models of succession have been applied to revegetation by Hammermeister (2001), Pitchford (2000) and Petherbridge (2000). While it is recognized that a multitude of successional models could be used to explain community development during revegetation (Hammermeister 2001), each of these researchers explained community development in terms of orderly development towards the predisturbance system. Although many different mechanisms of this change can be invoked, after a disturbance the system begins a more or less directional change towards recovery.

Some the results from this study lend themselves toward working within a traditional successional framework. As noted above, the Natural Recovery treatment though it was separated by ordination results from other treatments, did not behave in the discrete states typical of state and transition models. Rather, on the Solonetzic sites the Natural Recovery treatments seem to progress from the point of origin in a more or less linear fashion towards the Undisturbed treatment.

The compositional changes of the Natural Recovery treatment also suggest progression along lines of traditional ecological theory. Early in development annual weeds dominated the communities. In years four and five the prominence of

these annual weeds are declining and perennial forbs and grasses such as *Artemisia frigida* Willd. (pasture sage) and *Hordeum jubatum* L. (foxtail barley) are establishing. Climax grasses such as *Stipa comata* and *Agropyron smithii* are also becoming prominent. These observations are consistent with previous researchers who documented natural secondary succession on abandoned crop lands in the Great Plains. Judd (1940) and Judd and Jackson (1939) reported that the first few years of succession are dominated by annual weeds including *Salsola kali* and *Kochia scoparia*. The next stage of secondary succession is dominated by perennial forbs and grasses including *Artemisia frigida, Hordeum jubatum* and *Sitanion hystrix* (Nutt.) J.G. Smith (squirreltail). Both Judd (1940) and Tolstead (1941) reported *Agropyron smithii* and *Stipa comata* were among the climax grasses to establish early in revegetation. It would appear from species cover data in this study that Natural Recovery treatments are well on their way to establishing the *Agropyron smithii* communities reported in the early literature following cropland abandonment.

It seems apparent that the Natural Recovery treatments are following the expected pattern of secondary succession, however, it is less clear, how long it will take for the climax plant community to reestablish or even if it will. Reestablishment of MGP after cropping has been reported to take 20 years (Judd and Jackson 1939, Tolstead 1941), 30 years (Judd 1940) or longer (Costello 1944). Samuel and Hart (1994) reported that even after 61 years of secondary succession, climax plant communities did not reestablish in Wyoming MGP.

Plant communities quite different from climax types may also reestablish. Dormaar and Smoliak (1985) reported that abandoned cropland in MGP became a community dominated entirely by *Stipa comata* and was completely missing the key

climax species, *Bouteloua gracilis*. The failure of *Bouteloua gracilis* to establish on disturbed prairie has been attributed to the lack of viable seed and opportunities for establishment (Samuel and Hart 1994). However, in this study, *Bouteoula gracilis* had a reasonable amount of cover in the Natural Recovery in 1999 but decreased in 2000, likely due to the extreme drought. Since the Natural Recovery treatment seemed to follow the successional patterns seen in abandoned cropland, wellsites should recover to a near climax community within fifty years after disturbance. One of the primary factors affecting the recovery of abandoned cropland is the proximity of a native prairie seed source (Judd 1940, Shantz 1917). Given that wellsites are usually less than a one ha in size, adjacent native seed sources should be present.

Grazing did not affect the successional vectors of Natural Recovery treatments. Previous researchers have generally found that grazing retarded natural secondary succession on abandoned croplands (Costello 1944, Tomanek et al. 1955). Grazing had negative effects on the ecosystem function of the Natural Recovery treatment (Chapter V). While no pattern is evident in the data, field observations would suggest that grazing is retarding secondary succession of Natural Recovery treatment. One particular observation is the uprooting of small bunchgrasses by cattle. The Grazed Natural Recovery treatments often were littered with small *Hordeum jubatum* and *Sitanion hystrix* plants that were pulled up by the roots and left on the ground.

3.4 Predicting long term trends in wellsite revegetation

While both state and transition models and traditional succession have proven useful to explain the vegetation trends observed in the early stages of wellsite

revegetation, the ultimate goal of predictive ecology is to be able to predict future events. Based on successional trends determined from reciprocal averaging results, seed mixes that include wheatgrass cultivars will likely remain dominated by these species. Some external force may be required move them to a new domain. Grazing has potential to facilitate this transition by reducing the cover of *Agropyron dasystachyum*. Natural Recovery treatments will likely continue along the traditional path of succession. Recovery to the predisturbance plant community may take as long as 20 to 50 years. *Bouteloua gracilis* may take an even longer time to establish. Grazing may retard the development of the Natural Recovery treatment.

4.0 Conclusions

Both state and transition models and traditional successional theory were useful to describe community trends in the early revegetation of wellsites. The state and transition approach was useful to describe the effect of seeding and the competition from native wheatgrass cultivars. Traditional succession was useful to describe the development of the Natural Recovery treatment. Models used to explain observed community trends were strong enough to make predictions about future community trends. Treatments that included native wheatgrass cultivars will likely remain similar to their current state. Grazing may be a potential tool to break out of this state. Natural Recovery treatments will slowly progress toward a predisturbance community which may take as long as 50 years.

Table 4-1: Mean abundance of selected species on an eleven point cover scale in five revegetation treatments on four wellsites on Chernozemic soils in mixed grass prairie.

Species	Treatment	1997	1998	1999	2000
Agropyron dasystachyum	P Value	0.0001	0.0001	0.0001	0.0001
	Standard Error	0.18	0.44	0.60	0.32
	Current	2.32a[1]	3.18a	4.79a	2.76a
	Diverse	1.35b	1.83b	2.04b	1.85b
	Simple	1.64b	2.88a	4.53a	3.18a
	Natural Recovery	0.10c	0.06c	0.07c	0.12c
	Undisturbed	0.28c	0c	0c	0c
Agropyron smithii	P Value	0.03	0.02	0.15*	0.01*
	Standard Error	0.35	0.35	0.53	0.21
	Current	0.9ab	0.95a	1.87a	1.07ab
	Diverse	0.4bc	0.84a	1.75a	1.21a
	Simple	0.6abc	1.26a	1.32ab	1.15a
	Natural Recovery	0.02c	0.08b	0.57b	0.49c
	Undisturbed	1.4a	1.36a	1.01ab	0.68bc
Agropyron trachycaulum	P Value	0.001	0.0001	0.001*	0.003
	Standard Error	0.24	0.28	0.58	0.17
	Current	2.16a	2.09a	1.47b	0.57a
	Diverse	1.68a	2.11a	2.94a	0.68a
	Simple	0.03b	0b	0.14c	0.07b
	Natural Recovery	0b	0.01b	0.02c	0.60a
	Undisturbed	0b	0b	0c	0b
Bouteloua gracilis	P Value	0.0001	0.0001	0.0001	0.0001
	Standard Error	0.23	0.20	0.19	0.13
	Current	0.09b	0.01b	0.02b	0.01b
	Diverse	0.33b	0.23b	0.15b	0.19b
	Simple	0.42b	0.42b	0.16b	0.14b
	Natural Recovery	0.02b	0.04b	0.34b	0.08b
	Undisturbed	2.64a	2.37a	2.02a	1.07a
Kochia scoparia	P Value	0.41	0.32	0.42	0.21
	Standard Error	0.22	0.91	0.49	0.28
	Current	0a	0.58a	0.64a	0.21ab
	Diverse	0.14a	1.16a	0.95a	0.43ab
	Simple	0.07a	1.27a	0.44a	0.23ab
	Natural Recovery	0.13a	1.96a	0.72a	0.68a
	Undisturbed	0a	0a	0a	0b
Salsola kali	P Value	0.36	0.03	0.08*	0.16
	Standard Error	0.23	0.59	0.53	0.20
	Current	0.35a	0.81b	0.43ab	0.01b
	Diverse	0.15a	0.56b	0.10b	0b
	Simple	0.19a	0.93b	0.19b	0b
	Natural Recovery	0.41a	2.29a	1.47a	0.45a
	Undisturbed	0a	0b	0b	0b
Stipa comata	P Value	0.006	0.0001	0.0001*	0.0001
	Standard Error	0.61	0.35	0.37	0.26
	Current	0.10b	0.02b	0.06c	0b
	Diverse	0.58b	0.3b	0.50bc	0.16b
	Simple	0.47b	0.46b	0.75bc	0.19b
	Natural Recovery	0.09b	0.33b	1.02b	0.51b
	Undisturbed	3.43a	3.18a	3.83a	2.76a

[1] Means with different letters are significantly different within a year

*F-test recalculated using Proc Mixed

Table 4-2: Mean abundance of selected species on an eleven point cover scale in five revegetation treatments on three wellsites on Solonetzic soils in mixed grass prairie.

Species	Treatment	1997	1998	1999	2000
Agropyron dasystachyum	P Value	0.0005	0.002	0.0003	0.0001
	Standard Error	0.23	0.53	0.88	0.36
	Current	1.66a[1]	2.44ab	3.83ab	3.14a
	Diverse	0.95b	1.78bc	3.53b	2.67a
	Simple	1.69a	3.43a	5.28a	3.20a
	Natural Recovery	0.04c	0.06d	0.14c	0.03c
	Undisturbed	0.90b	1.12cd	0.55c	1.47b
Agropyron smithii	P Value	0.004	0.007	0.01	0.02
	Standard Error	018	0.23	0.31	0.28
	Current	0.91ab	1.36a	1.94a	1.47a
	Diverse	0.57b	1.06a	1.67a	1.45a
	Simple	1.04a	1.05a	1.53a	1.14ab
	Natural Recovery	0.06c	0.10b	0.41b	0.50bc
	Undisturbed	0.90ab	0.85a	1.28a	0.47c
Agropyron trachycaulum	P Value	0.0001	0.04	0.002	0.002
	Standard Error	0.35	1.10	0.59	0.20
	Current	2.68a	2.62a	2.25a	1.14a
	Diverse	2.08a	3.45a	3.19a	0.89ab
	Simple	0.22b	0b	0b	0.17cd
	Natural Recovery	0.02b	0b	0.53b	0.53bc
	Undisturbed	0b	0b	0.06b	0.03d
Bouteloua gracilis	P Value	0.04	0.001	0.17	0.12
	Standard Error	0.69	0.22	0.36	0.22
	Current	0.11b	0.02b	0.03b	0b
	Diverse	0.25b	0.27b	0.11ab	0.17ab
	Simple	0.22b	0.37b	0.28ab	0.17ab
	Natural Recovery	0.03b	0.16b	0b	0b
	Undisturbed	1.52a	1.44a	0.70a	0.58a
Kochia scoparia	P Value	0.44	0.40	0.46	NA
	Standard Error	0.51	1.18	0.52	0
	Current	0.01a	0.17a	0.25a	0a
	Diverse	0.38a	1.77a	0.14a	0a
	Simple	0.87a	1.37a	0.94a	0a
	Natural Recovery	0.51a	1.92a	0.50a	0a
	Undisturbed	0a	0.02a	0a	0a
Salsola kali	P Value	0.08	0.11	0.10	0.46
	Standard Error	0.11	0.22	0.12	0.02
	Current	0.03b	0.12b	0.08ab	0a
	Diverse	0.08ab	0.26ab	0b	0a
	Simple	0.06ab	0.48ab	0.06b	0a
	Natural Recovery	0.13a	0.72a	0.31a	0.03a
	Undisturbed	0b	0.10b	0b	0a
Stipa comata	P Value	0.0008	0.0006	0.002	0.0005
	Standard Error	0.22	0.16	0.28	0.23
	Current	0.14b	0.04b	0.03b	0c
	Diverse	0.22b	0.35b	0.56b	0.28bc
	Simple	0.28b	0.41b	0.22b	0.08c
	Natural Recovery	0.10b	0.19b	0.39b	0.72b
	Undisturbed	1.51a	1.31a	1.81a	1.64a

[1] Means with different letters are significantly different within a year

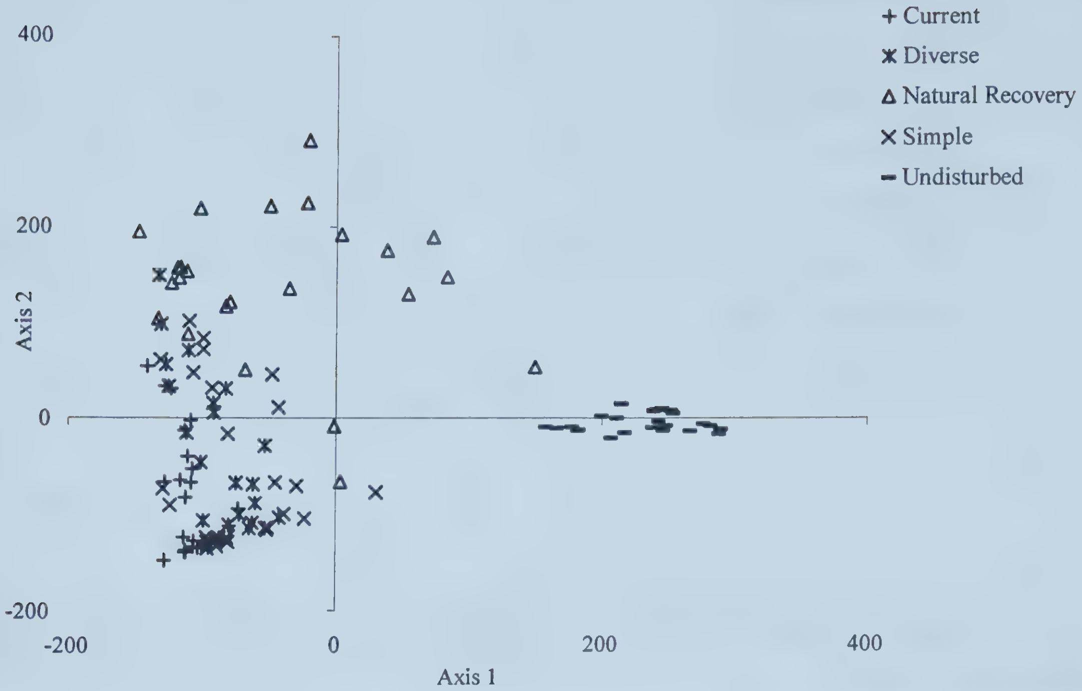

Figure 4-1: Reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Chernozemic soils 1997 to 2000.

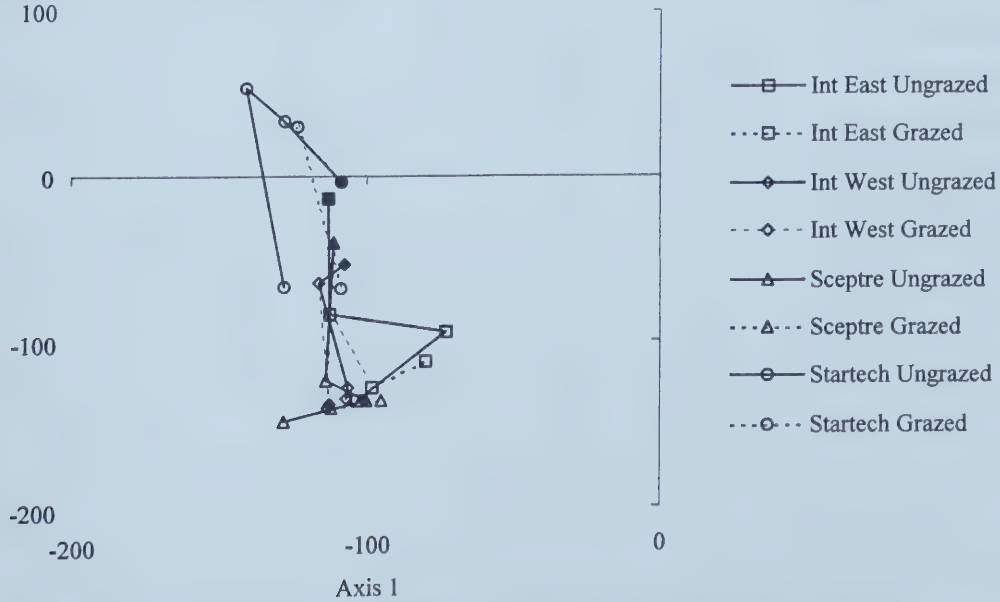

Figure 4-2: Vectors of Current treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Chernozemic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

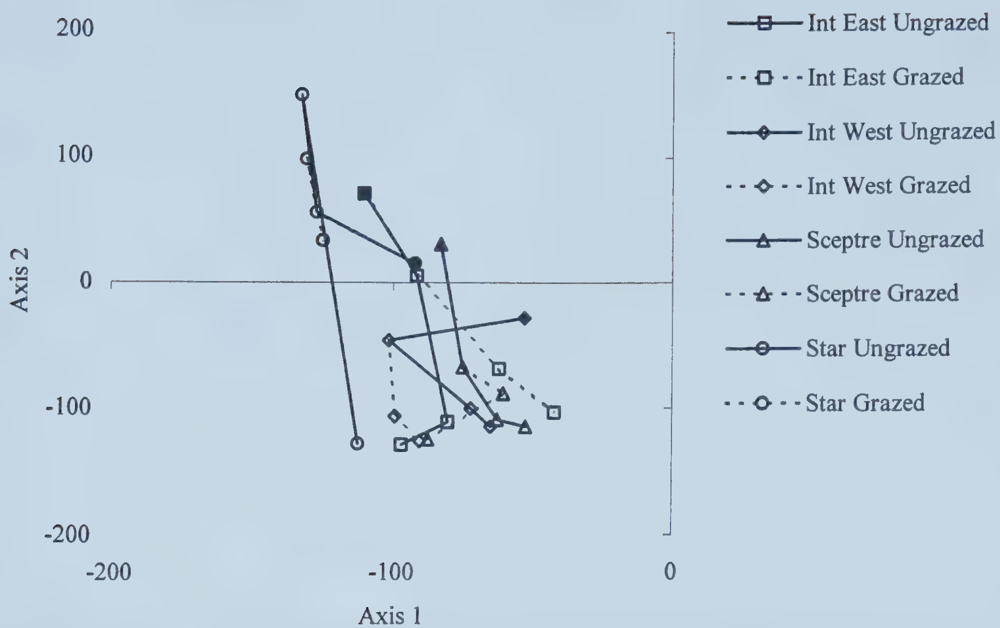

Figure 4-3: Vectors of Diverse treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Chernozemic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

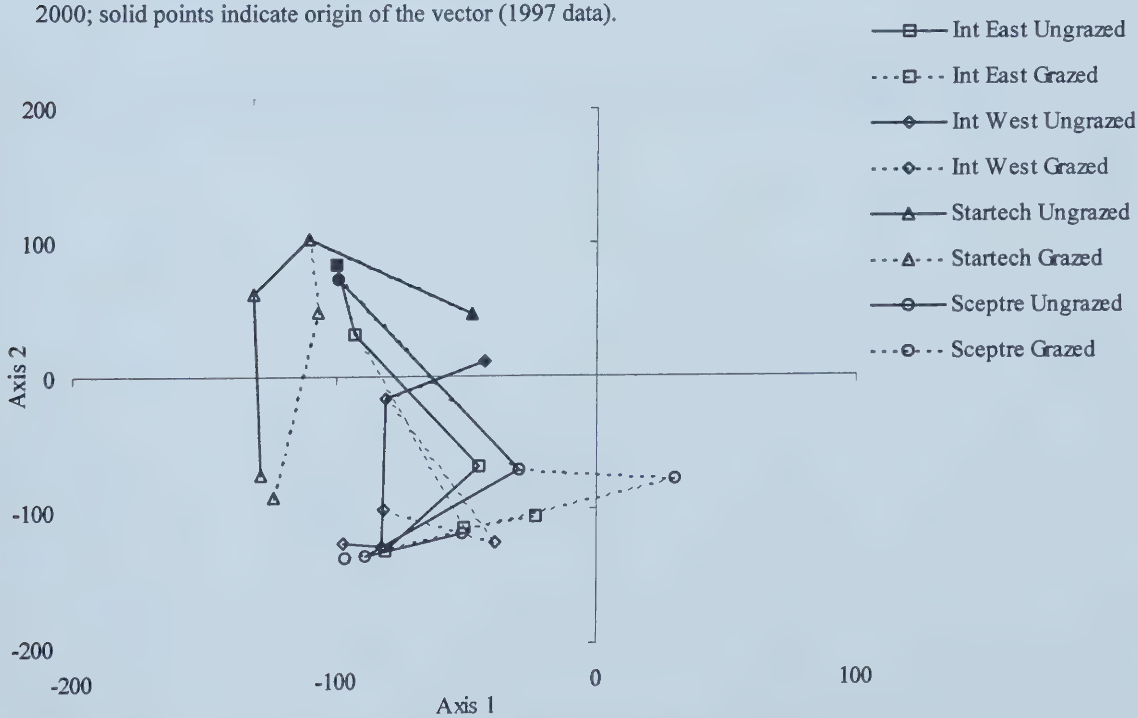

Figure 4-4: Vectors of Simple treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments four wellsites on Chernozemic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

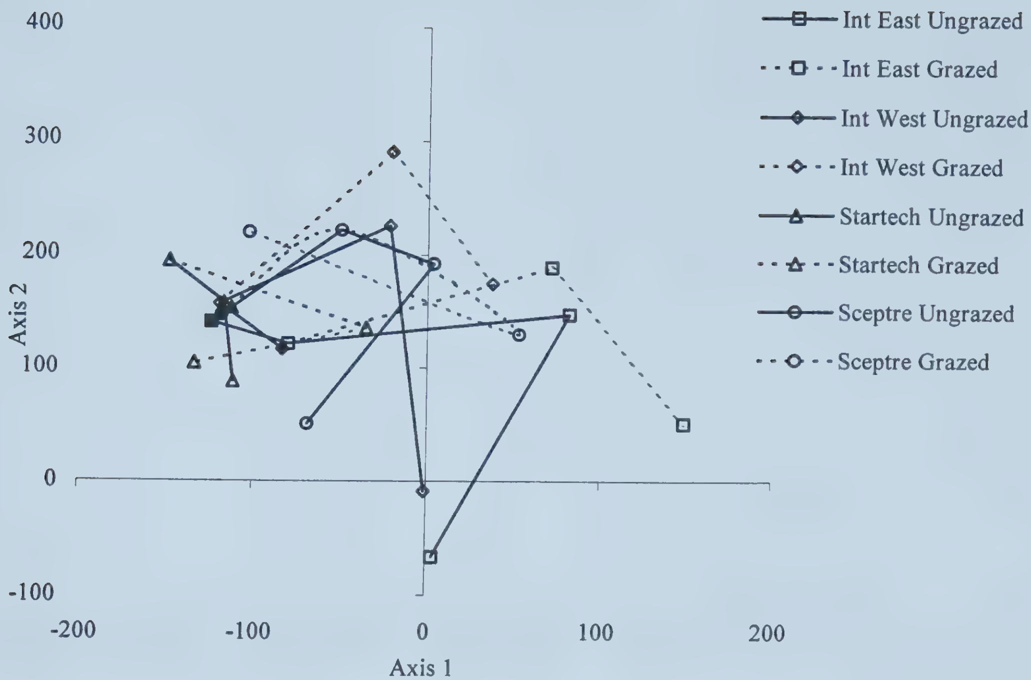

Figure 4-5: Vectors of Natural Recovery treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Chernozemic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

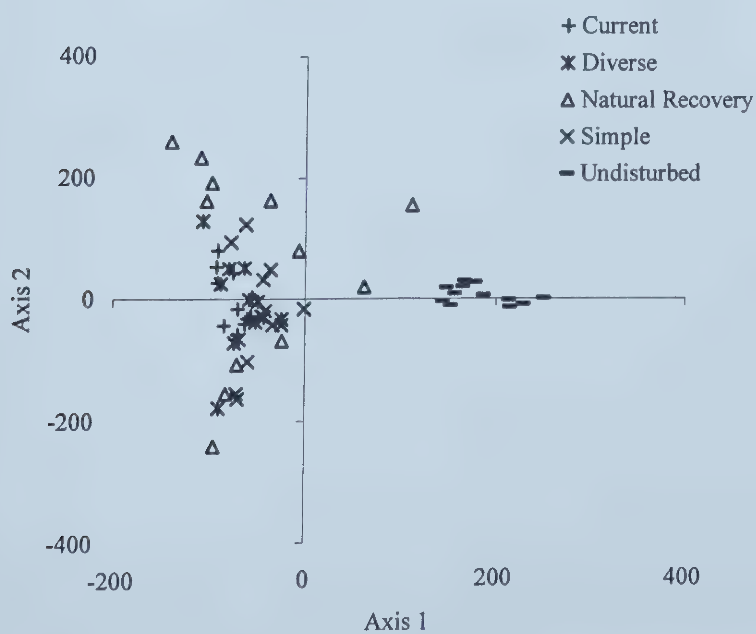

Figure 4-6: Reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Solonetzic soils 1997 to 2000.

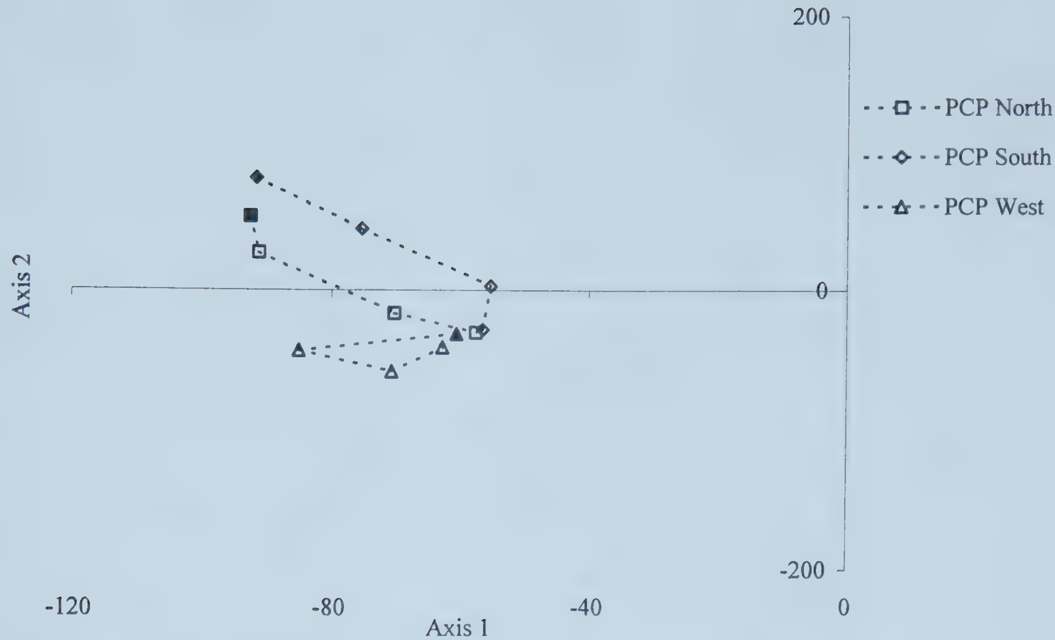

Figure 4-7: Vectors of Current treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Solonetzic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

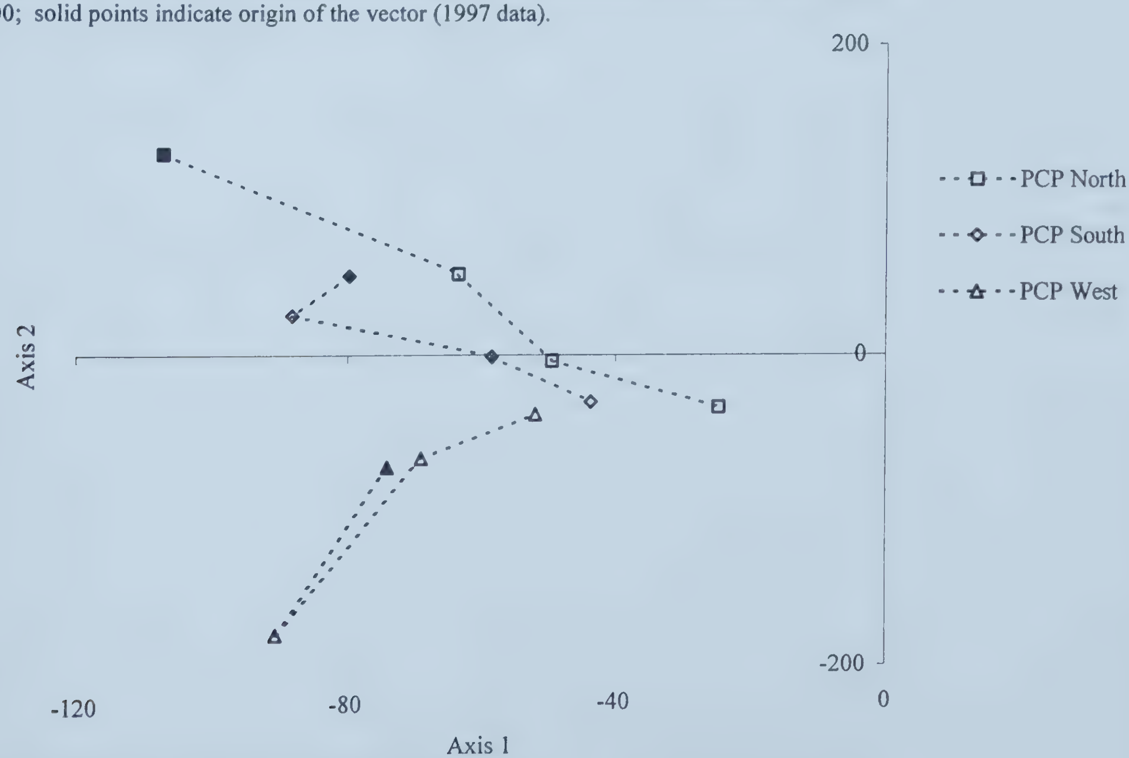

Figure 4-8: Vectors of Diverse treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Solonetzic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

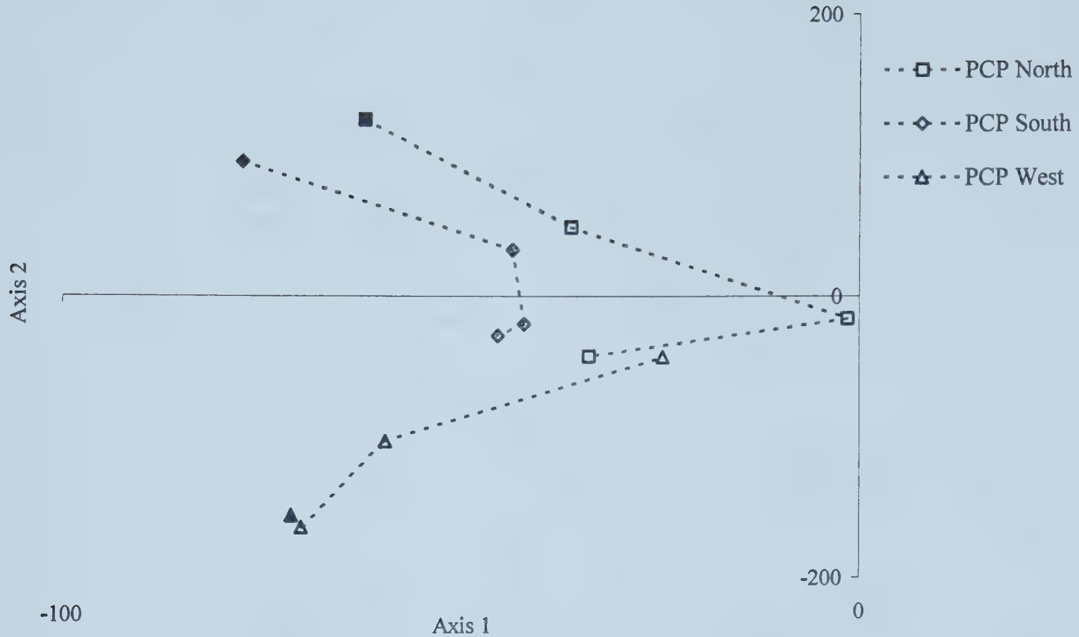

Figure 4-9: Vectors of Simple treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Solonetzic soils 1997 to 2000; solid points indicate origin of the vector (1997 data).

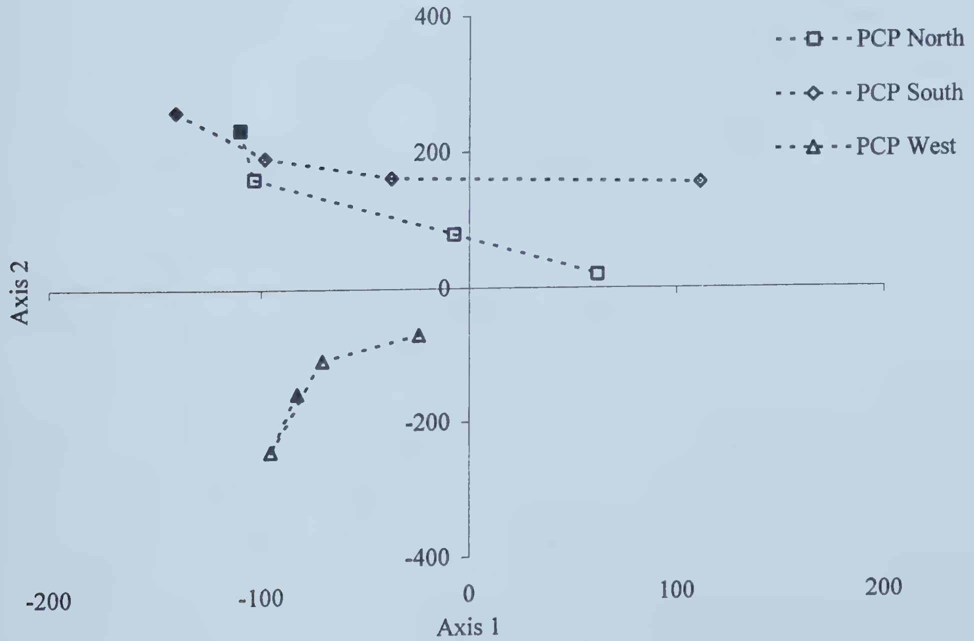

Figure 4-10: Vectors of Natural Recovery treatments from a reciprocal averaging ordination of species composition data from five revegetation treatments on four wellsites on Solonetzic soils 1997 to 2000; solid points indicate origin of the vector (1997 data)

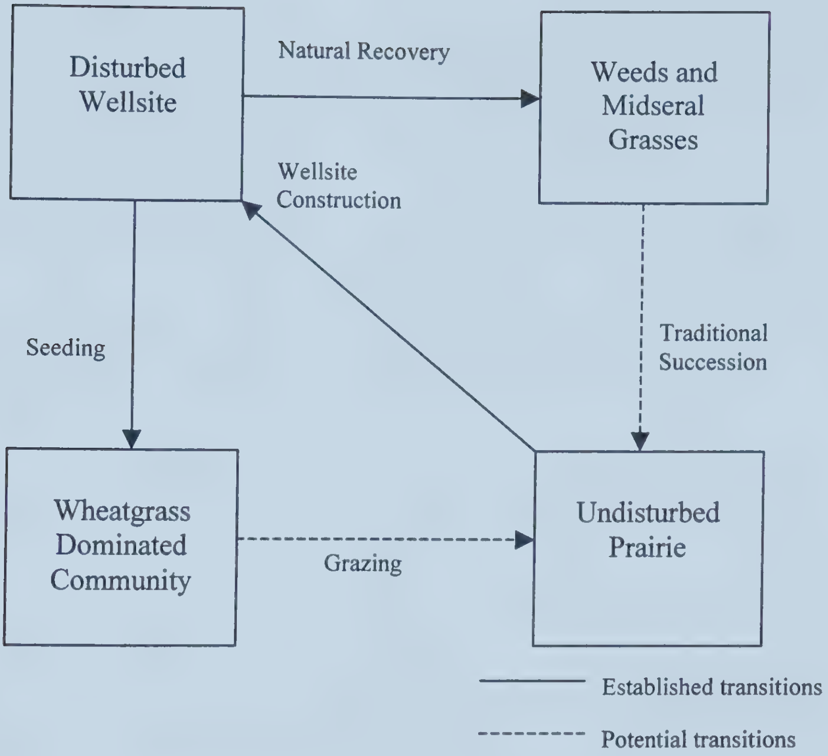

Figure 4-11: State and transition model of mixed grass prairie revegetation after wellsite disturbance.

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Chapter V Ecosystem Function and Mixed Grass Prairie Wellsite Revegetation

1.0 Introduction

Restoration activities often focus on the scale of ecosystem function. Rather than the traditional focus of reestablishing plant and animal communities, coarser patterns of ecosystem function are the object of interest (Bradshaw 1987). Ecosystem characteristics such as nutrient cycling, productivity, and stability are emphasized at this scale (Ehrenfeld and Toth 1997). Practitioners who work with drastically disturbed lands have usually advocated this approach (Bradshaw 1983). However, the approach may be useful in a wider variety of restoration activities.

Aronson et al. (1993) suggested a set of "vital ecosystem attributes" could be identified for measuring restoration success in arid and semiarid lands. These attributes could be classified as structural (cover, diversity, functional composition) and functional attributes (productivity, nutrient cycling). The theoretical framework for this approach is the "strategy of ecosystem development" proposed by Odum (1967). As ecosystems develop over time or recover from disturbance, biomass accumulates, productivity increases, diversity increases and in general, ecosystems become more complex and stable. This approach has developed as a tradition in ecology distinct from community and population ecology (O'Neil et al. 1986). By focusing on coarse scale patterns of material and energy flow, ecosystem approaches may offer unique insight into restoration practice (Ehrenfeld and Toth 1997).

Naeth (1985) and Hammermeister (2000) have applied the ecosystem approach to restoration of MGP in Alberta. Both of these researchers investigated the effects of petroleum development on MGP. Oil and gas wellsite construction is one such common development. Over 30,000 active and 10,000 abandoned wells were reported to be in MGP of Alberta in 1991 (Kerr et al. 1993). Wellsite construction usually involves the destruction of vegetation and stockpiling of topsoil and subsoil on an area 100 x 100 m (Kerr et al. 1993). Current industry restoration practices include seeding aggressive wheatgrass cultivars and exclusion of cattle grazing for up to five years after seeding. Alternative restoration practices involve including more species in seed mixes, not seeding (Natural Recovery) and allowing cattle grazing to occur earlier in the revegetation process. The necessity of managing restoration projects has been widely recognized (Wark et al 1996) and cattle grazing presents an obvious management option for MGP wellsites.

This study investigated the impact of these alternative revegetation techniques on native prairie from an ecosystem function perspective. Structural attributes (cover, species evenness, species richness) and functional attributes (biomass production, soil nitrogen, light levels) were monitored and the "vital ecosystem attributes" framework of Aronson et al. (1993) used to evaluate restoration success.

2.0 Methods

2.1 Site description

The study took place in seven sites of native prairie in southeastern Alberta, Canada from May 1999 to August 2000 (Figure A-1). Two sites were located 300 m

apart, approximately 20 km northwest of Medicine Hat on loam textured Orthic Brown Chernozemic soils. Two sites were 3 km apart, 15 km northeast of Bow Island on sandy loam textured Orthic Brown Chernozemic soils. Three sites on Solonetzic sites were located within 50 km of Brooks. Natural vegetation in this area is referred to as dry MGP (Strong and Leggat 1992) or MGP (Coupland 1961). Undisturbed prairie at each of the sites was dominated by *Stipa comata* Trin & Rupr. (needle and thread) and *Bouteloua gracilis* (HBK) Lag. (blue grama grass). *Agropyron dasystachyum* (Hook) Scubn (northern wheatgrass) is also a dominant grass on Solonetzic soils. Taxonomy follows Moss (1983). The climate is semiarid and the majority of growing season precipitation falls in June. May to September precipitation was slightly below long-term averages in 1996 to 1998 and moderately above long-term averages in 1999 (Table A-1). In 2000 precipitation was approximately half the long-term average.

Each of the research sites was a drilled and abandoned wellsite located inside a field of native prairie used for grazing. Cattle have grazed each of the fields for over thirty years. In 1999, the range condition of the sites was estimated to vary from good to excellent (Wroe et al. 1985).

Wellsites were established in either fall 1995 or spring 1996 and abandoned in spring 1996. Each of the companies contributing wellsites was asked to reclaim them using normal practices up to the point of being seeded. Each wellsite had the topsoil and subsoil separately removed and stockpiled. After industrial activities were finished, topsoil and subsoil were replaced and a seedbed prepared. Each wellsite and

an adjacent strip of undisturbed native prairie were fenced to exclude grazing for the first three years of the study.

2.2 Treatments and experimental design

Three seeded and one non-seeded (Natural Recovery) treatments were compared to the Undisturbed treatment in this study. The seeded treatments consisted of a three species, wheatgrass dominated mix (Current), a five species mix (Simple) and a 21 species mix (Diverse) (Table A-2). Each of these treatments was investigated with and without cattle grazing on Chernozemic sites.

Each of the wellsites was treated as a block. The 3 seeded and Natural Recovery treatments were placed in the same location within each block because of the influence of predominant wind direction on natural revegetation (Table A-2). The position of the Undisturbed treatment varied with each block. Exclosures allowed comparison of Ungrazed and Grazed areas within each revegetation treatment.

2.3 Treatment application

Revegetation treatments were implemented in spring 1996 by Hammermeister (2001). Seeding rates (kg/ha) were calculated based on a desired number of pure live seeds (PLS) per unit area (300 PLS/m^2). Because purity and viability data were not available for the forbs, they were estimated as 66%. Chick starter was used as a carrier to ensure even seed distribution and improve seed flow. The total seeding rate was applied with two perpendicular passes of a calibrated Truax native seed drill. *Koeleria macrantha* (Ledeb.) J.A. Schultes f. (june grass) was broadcast with chick

starter following seeding but before straw crimping. Although the seeder was calibrated, the seeding rate varied slightly due to limited accuracy of openers (10%).

During summer 1999 grazing exclosures were built in the center of each Chernozemic site enclosing part of each revegetation treatment. The unfenced portion of each treatment was grazed in June and July 1999. Grazing treatments were applied by moving cattle from the adjoining field into the fenced wellsite where the animals were free to graze any of the treatments outside the exclosure. Water was provided in troughs placed directly beside the gate of each wellsite. Cattle remained in the wellsite until visually estimated utilization approached 30 to 50% on each treatment. A confounding factor to the experimental design is that two of the sites were mowed two weeks prior to the application of the grazing treatment in 1999. Because of this complication, stocking rates of the Grazing treatment were reduced for these two sites (Table A-3). In summer 2000, mowed and unmowed sites were visually similar and all blocks were included in the analysis. Summer 1999 represented a period of above average precipitation while 2000 was below average precipitation so stocking rates were reduced. No grazing treatments were applied to Solonetzic sites.

2.4 Data collection

In 1997 and 1998 cover data were collected from thirty 0.1 m^2 quadrats placed at even intervals along three parallel transects in each treatment (Hammermeister 2001). Absolute (non-relativized) canopy cover was estimated to the nearest percent for each species. Percent bareground, litter cover and plant basal cover were also estimated to a total of 100 percent. Live percent cover was calculated for 1997 and 1998 by totaling the cover estimates of each species (Chapter IV). Because of these

data collection methods, live cover, litter cover and percent baregound do not total 100 for 1997 and 1998. Biomass production was determined by clipping vegetation to 2.5 cm height in three randomly placed 0.1 m^2 quadrats along each transect. Vegetation was oven dried at approximately 60 °C and weighed.

In 1999, grazing exclosures were built on the Chernozemic sites and so a different sampling scheme was adopted in 1999 and 2000. While exclosures were not present on Solonetzic sites, the sampling scheme was also changed to keep procedures consistent. In each plot one corner was arbitrarily chosen for the starting point. At three random distances from the corner, along one side of the plot transects were run. Each transect ran perpendicular to the side of the plot. At three (Chernozemic sites) or six (Solonetzic) m intervals along each transect, four 0.1 m^2 quadrats were located for a total of twelve quadrats in each plot (treatment combination). Species area curves conducted on 1997 data indicated that twelve quadrats was sufficient to describe communities. Species cover was estimated along an eleven point cover scale (1 = present to 5%, 2 = 6 to 10%, 3= 11 to 20% 4 = 21 to 30%, 5= 31 to 40%, 6= 41 to 50%, 7= 51 to 60%, 8= 61 to 70%, 9= 71 to 80%, 10= 81 to 90%, 11= 91-100%) in each quadrat along with additional estimates of percent bareground, ground litter, standing litter, cryptogram and live vegetation cover. These four components always totaled 100%.

Biomass production was assessed in 1999 and 2000 by clipping three randomly placed 0.25 m^2 quadrats in each plot. This number of quadrats constituted the greatest number that could be clipped given the field time available. Vegetation was clipped to the crowns (1 to 2.5 cm above soil) and coarsely sorted in standing

litter (brown vegetation which could be loosely pulled from the sward) and live vegetation (green and yellow). Ground litter was also hand raked and included all residual biomass above the mineral soil level including remnant crimped straw. *Selaginella densa* Rydb. (little club moss) was not included in any biomass samples. In 1999, ground and standing litter were not separated. Samples were oven dried at approximately 60 °C and weighed.

Soil subsamples to a 10 cm depth were collected in late summer at random locations along each of the transects used for vegetation sampling. Three, eight and nine soil samples were collected per plot in 1997, 1998 and 1999/2000, respectively. All soil subsamples were treated as a composite and a small subsample air dried and hand sieved through a 2 mm mesh to conduct analysis. Mineral nitrogen was determined by 2 *M* KCl extractable NO_3^- and NH_4^+ (Bremner, 1965).

Available light measurements were made in each plot in 1999 and 2000. In each plot one corner was arbitrarily chosen for the starting point. Transects were run at three random distances from the corner along one side of the plot transects were run. Each transect ran perpendicular to one side of the plot. At three (Chernozemic sites) or six (Solonetzic) m intervals along each transect, three light readings were made for a total of nine readings in each plot (treatment combination). Light readings were taken at ground level and 10 to 20 cm above vegetation by use of a standard greenhouse light meter. Available light was calculated by dividing ground level light with above vegetation light. The nine readings were averaged to give one value for each plot. Light readings were taken between 1000 and 1500 h but light conditions

varied from full sunlight to overcast. Light readings were not collected in 1997 and 1998 and one of the Chernozemic blocks in was missed in 1999.

2.5 Data analyses

For all data, subsamples (if collected) were averaged to give one value for each plot. Species cover estimates from 1997 and 1998 were converted to the 11 point cover scale used in 1999 and 2000. Species richness were calculated for each plot from species cover data using PC-Ord software (McCune and Mefford 1997). Data were analyzed separately for each year and each soil type. Data are presented in chronological order for interpretation of trends through time. Solonetzic sites were analyzed as a randomized complete block design using analysis of variance (ANOVA). Chernozemic sites were analyzed as a randomized complete block design for 1997 and 1998 and as a strip plot for 1999 and 2000. Treatment means were compared using the least significant difference (LSD) at alpha equal to 0.05. SAS software (SAS Institute Inc. 2000) was used to analyze the data using Proc GLM. Proc GLM uses the wrong error terms for split designs and so correct F-tests obtained by use of a random statement and specifying the error term in mean comparison tests. Inspection of mean sums of squares revealed that in some cases the residual error was larger than the block by treatment interactions (error terms A and B). To find correct error terms Proc Mixed was used to calculate F-tests and conduct mean comparison tests. In cases where interactions had to be investigated, Proc Mixed was used to investigate comparison of least squared means.

Given the small sample size, tests were not made for normality or homogeneity of variance. AVOVA is generally recognized as robust to violations of

both of these assumptions (Day and Quinn 1989). The third assumption of ANOVA, independence of samples, is also violated in this analysis by the lack of randomization. The fixed location of treatments in this study increases the risk of a Type I error. However many precedents exist to justify this sort of analysis in field studies and choice of analysis should reflect the judgement and objectives of the researcher rather than inflexible statistical rules of thumb (Stewart-Oaten 1995).

3.0 Results and Discussion

3.1 General results

Generally, live vegetation cover was 50% or greater in all treatments and years except 2000 during a drought (Table 5-1). In general either no difference existed among treatments or the Natural Recovery treatment had lower live cover. On Chernozemic sites, grazing significantly reduced live cover in 2000 ($p=0.01$). Ungrazed treatments had a live cover of 28% while Grazed treatments had a live cover 19% (standard error = 1.1) In 1999, significant revegetation treatment by block interactions were present.

In seeded and Natural Recovery treatments, ground litter cover was low in 1997, increased in 1998 and seemed to stabilize at 20 to 30% in 1999 and 2000. On Chernozemic sites, grazing significantly increased ground litter cover in 2000 ($p = 0.015$). Ground litter in Grazed treatments was 41% while in Ungrazed treatments it was 27% (standard error =2.7).

In seeded revegetation treatments, bareground averaged 30 to 50% in 1997 and 1998 and decreased to 10% or lower in 2000. The Undisturbed treatment had

significantly lower bareground than other revegetation treatments in 1997 and 1998. In 2000, the Natural Recovery treatment had significantly more bareground than other treatments on Chernozemic sites. Grazing significantly increased bareground in seeded and Natural Recovery treatments but not the Undisturbed. Grazing increased percent bareground in the Current treatment from 5.8 to 17.4%, from 15.3 to 7.5% in the Diverse treatment, from 17.0 to 44.5% in the Natural Recovery treatment and from 6.7 to 18.1% in the Simple treatment (standard error = 4.1).

Species richness was highest in 1997 and 1998 and declined in 1999 and 2000 (Table 5-2). In general, the Diverse treatment had the highest species richness among seeded treatments while after 1997, the Current treatment had the lowest. Excluding the Undisturbed treatment, the Natural Recovery treatment had the highest richness in 1999 and 2000. Grazing did not significantly change species richness.

Total above ground biomass in seeded and Natural Recovery treatments was around 50 g/0.1m^2 in 1999 and 2000 (Table 5-3). Generally, the Undisturbed treatment had significantly less above ground biomass than seeded and Natural Recovery treatments. The Natural Recovery treatment had the lowest above ground biomass among seeded and Natural Recovery treatments, however, only on Chernozemic sites in 2000 was this difference statistically significant. Grazing did not have a significant effect on total above ground biomass.

Live above ground biomass in seeded and Natural Recovery treatments averaged 20 g/0.1 m^2 in 1997, 1998 and 1999 and 15 g/0.1 m^2 in 2000. Live biomass was significantly lower in the Undisturbed treatment averaging around 10 g/0.1 m^2. Grazing did not have a significant impact on live above ground biomass.

Above ground litter biomass on seeded and Natural Recovery treatments averaged around 30 to 50 $g/0.1m^2$. The Undisturbed treatment had significantly less litter than seeded and Natural Recovery treatments except on Solonetzic soils in 2000. Grazing did not have a significant impact on litter, however when ground litter was analyzed as a separate variable (results not shown), grazing significantly increased the amount. Ground litter was 35 $g/0.1m^2$ on Grazed treatments and 26 $g/0.1m^2$ on Ungrazed treatments (standard error = 2.2).

On Chernozemic soils nitrate levels generally did not differ among revegetation or grazing treatments (Table 5-4). On Solonetzic soils the Natural Recovery treatment had significantly higher nitrate levels than all other treatments in 1999 and 2000. Similarly, ammonium levels did not differ among revegetation or grazing treatments. Total soil nitrogen was not affected by revegetation treatment on Chernozemic soils but on Solonetzic sites, levels were significantly higher in the Natural Recovery treatment in 1999 and 2000. Grazing did not significantly affect total nitrogen levels.

The Natural Recovery and Undisturbed treatments generally had higher light levels than other revegetation treatments (Table 5-5). Grazing by revegetation treatment interactions were significant on Chernozemic sites in 2000. Grazing significantly increased light levels in all treatments except the Undisturbed in 2000. Grazing increased percent light in the Current treatment from 0.49 to 0.82, from 0.50 to 0.78 in the Diverse treatment, from 0.48 to 0.92 in the Natural Recovery treatment and from 0.45 to 0.76 in the Simple treatment (standard error = 0.05).

3.2 Cover components

One component of ecosystem structure is the physical arrangement of objects in space. Percent cover estimates were used to measure the physical characteristics of the grass swards in this prairie ecosystem. All seeded revegetation treatments were structurally similar to the Undisturbed treatment in terms of live cover, litter cover and bareground. The Natural Recovery treatment was structurally different from the other treatments by having more bareground, usually less live cover and in some cases, less litter cover. Bareground is an important structural component of grassland ecosystems. The amount of exposed bareground gives an indication of the system's capacity to resist soil erosion. A threshold of 25 to 50% bareground is often cited as the level at which serious soil erosion could occur (Weltz et al. 1998). The structural attributes of revegetated wellsites in MGP in this study should provide adequate protection against such erosion. The exception to this may be the Natural Recovery treatment that had near threshold levels of bareground.

Previous researchers determined that cover components of grasslands play important functional roles in the ecosystem. Litter provides important roles in preventing evapotranspiration and aiding infiltration (Naeth et al. 1991a). Vegetative cover was considered one of the "vital ecosystem attributes" of Aronson et al. (1993) and is probably the most common measure of success in restoration that seeks to reestablish ecosystem function. Evidence from this study indicates that each of the seeded and Natural Recovery treatments was capable of reestablishing cover equal to the predisturbance community. No one seeding treatment appeared to perform better than others at establishing cover components.

Grazing also impacted structural components of the ecosystem. Grazing significantly reduced live cover and increased ground litter cover. We can then infer that grazing removed live material but also increased the amount of dead plant material on the ground through trampling. Apparently the increase of ground litter was not adequate to compensate for the loss of canopy material, because bareground was increased by grazing. On the Natural Recovery treatment this increase may be detrimental to the functioning of the system. Bareground on Grazed Natural Recovery treatments was 44%, posing a potential risk for soil erosion as it may exceed the threshold (Weltz et al. 1998).

Previous investigators have examined the role of grazing on cover components of grasslands. Naeth et al. (1991a) found that grazing increased the amount of baregound and decreased litter and live cover. Cattle grazing would appear to have important structural impacts on MGP wellsite disturbances.

3.3 Diversity

Diversity measured in this study as species richness is recognized in the literature as an important attribute of grassland ecosystems. Diversity is recognized as potentially important for nutrient cycling, productivity and ecosystem stability (Tilman et al. 1997). Overall, diversity and species richness of revegetated wellsites was similar to the Undisturbed treatment. The Current treatment was the least effective at establishing more species rich community. These results would suggest that there is some utility in using a more species in a seed mix to establish a diverse community. However, it is interesting to note that in 2000, the seeded treatments all had significantly lower species richness than the Undisturbed treatment. In contrast,

the Natural Recovery treatment tended to have the closest richness to the Undisturbed treatment. In the long-term the Natural Recovery treatment may be the most effective to establish a species rich and diverse community.

The importance of seed mix richness for restoration has been investigated by Bush (1997), Howat (1998), Pitchford (1999) and Hammermeister (2001). While seed mix richness generally impacted the diversity of revegetated communities, it was recognized that simply adding a species to the seed mixture did not mean it would be added to the plant community. Individual species characteristics such as competitive ability, germination requirements and growth rates probably play a more important role in the diversity of revegetated grasslands than simply the number of species available (Tilman et al. 1997). Tilman (1997) manipulated species richness in revegetated grasslands and found that diverse plant communities resisted invasion better than species poor mixes.

3.4 Biomass production

Biomass production is a key functional attribute for assessing success of restored grasslands. Biomass production indicates nutrient cycling, lack of impediments to plant growth and soil quality. Above ground net primary production (ANPP), collected in this study as live vegetation, is a good indicator of the health of the system (Aronson et al. 1993). ANPP values reported for the Undisturbed native prairie (3.5 to 11.1 $g/0.1m^2$) are similar to those reported for northern MGP (Willms et al. 1993). The NPP of the seeded treatments was usually much higher (20 $g/0.1m^2$). This high production is likely a result of aggressive wheatgrass cultivars and release of nitrogen from the wellsite disturbance (Hammermeister 2001).

It is interesting to note the variable response of the treatments to the unusually wet year in 1999 and the drought year in 2000. With abundant precipitation in 1999 the Undisturbed treatment nearly doubled typical ANPP. The seeded treatments, however, did not have such a large increase in ANPP. The dry conditions of 2000 resulted in a drop in ANPP within the Undisturbed treatment to average levels of production. In contrast, the seeded and Natural Recovery treatments dropped production to near half of usual levels. The Current treatment did not even have statistically higher productivity than the Undisturbed treatment. The more species rich Diverse and Simple treatments however had higher levels of production. These responses to the 2000 drought would suggest that more diverse systems are more stable (resilient to disturbance), the Undisturbed treatment may be the most long term stable community. These results are similar to Tilman and Downing (1994) who compared the stability of diverse and simple communities in 1987 following drought.

While ANPP (live biomass) is an important indicator of ecosystem function, accumulation of litter is required to ensure nutrients are returned to the soil. Litter in MGP has been recognized as important for decomposition (Redmann and Abouguendia 1978, Redmann 1978) and soil water (Naeth et al. 1991b). In this study, the amount of litter in the Undisturbed treatment varied between 3.1 to 17.5 g/0.1 m^2. These values are similar to other studies that have reported 6 to 12 g/0.1 m^2 of litter in MGP (Smoliak 1965, Willms et al. 1986, Willms et al. 1993). Litter levels in the seeded and Natural Recovery treatments were larger than Undisturbed (30 g/0.1 m^2). The straw crimp applied to the seeded and Natural Recovery treatments likely contributed to this greater litter level. It would appear that sufficient litter

accumulates in the seeded and Natural Recovery treatments to allow for decomposition and hydrological function. Grazing did not alter litter biomass as it did litter cover, however it did significantly increase ground litter biomass. This would suggest that the impact of grazing on seeded and Natural Recovery treatments is structural rather than functional. Cattle trampling physically moved more litter onto the soil surface. Ultimately this may result in more rapid decomposition and cycling of nutrients. Schuman et al. (1990) similarly found that cattle grazing on reclaimed mine land in Wyoming increased litter cover and may have improved ecosystem function.

Total above ground biomass is an important component of grassland ecosystems. Odum (1967) proposed that accumulation of biomass was one of the characteristics of a well developed ecosystem. In all seeded and Natural Recovery treatments above ground biomass was twice or more that of the Undisturbed treatment. All of the seeded and Natural Recovery treatments investigated would appear to be functionally mature by Odum's criteria, but also structurally different from the Undisturbed prairie.

3.5 Mineral nitrogen

Nitrate levels in the MGP are expected to increase with wellsite disturbance, then decrease with progressing succession (Johnston et al. 1967). Ammonium levels and the NH_4/NO_3 ratio are expected to increase over time after disturbance (Li and Redmann 1992) due to inhibition of nitrification (Rice and Pancholy 1972). This study did not find such a pattern. Nitrate levels did not differ between treatments on Chernozemic sites except in 1997. Hammermeister (2001), however, performed

more intensive sampling on the same sites in 1997 and 1998 and found that nitrate levels were higher on the disturbed seeded than Natural Recovery treatments. This study did find that on Solonetzic sites, the early successional Natural Recovery treatment had higher levels of nitrate than all other treatments in 1999 and 2000. Hammermeister (2001) argued that nitrate levels were reduced under seeding treatments due to uptake from perennial wheatgrasses or possibly allelochemicals that inhibit nitrification. While nitrate levels were decreasing, ammonium levels were not different among revegetation treatments. This would suggest that the NH_4/NO_3 ratio should be greater in the seeded treatments and Undisturbed treatment than in the Natural Recovery treatment consistent with sucessional theory (Rice and Pancholy 1972). The mineral nitrogen status of treatments investigated in this study therefore appears to be functioning to permit ecosystem reestablishment.

Several researchers have suggested that since early successional grasslands are nitrate rich, soil impoverishment may improve restoration success (Wilson and Gerry 1995, Jackson 1999). Fertilization experiments in MGP frequently leads to establishment of weedy species (Johnston et al. 1967). In this study, establishment of perennial grasses was sufficient to lower nitrate levels in MGP. It is unlikely that soil impoverishment would have substantial benefit to the revegetation of MGP wellsites.

3.6 Available light

Light levels have been identified as an important limiting factor in succession. Tilman (1985) has emphasized the importance of the ratio of light to soil resources as an important regulating factor in species replacement dynamics. Light levels are predicted to be high in early successional communities and lower in late successional

communities where light competition becomes more important. In this study, the early seral, Natural Recovery community light levels were high while those in the seeded treatments were low due to the high biomass accumulation from the aggressive wheatgrass cultivars. In the Undisturbed treatment composed of less aggressive species, light levels were also high. On revegetated MGP wellsites, light resource trends do not reflect the pattern predicted by Tilman (1985), however light levels are probably important for understanding the future of the system. Less dominant grasses are declining in the seeded treatments potentially due to light competition. Grazing increased light levels in the seeded treatments and thus may aid the establishment of less dominant grasses in the long term. Increasing light levels in the Natural Recovery treatment, in contrast, may allow persistence of weedy species and reduce mid-seral grasses.

3.7 Synthesis of vital ecosystem attributes

A variety of ecosystem attributes were examined in this study in the framework proposed by Aronson et al. (1993). The many attributes examined make it possible to characterize the ecosystem function of MGP wellsite revegetation treatments. Undisturbed MGP is typically a diverse, stable system with low bareground, low nitrate levels, high light levels and low above ground biomass. Natural Recovery treatments differ from the Undisturbed by having higher bareground and higher nitrate levels. Seeded treatments differ from the Undisturbed by having high above ground biomass, low light levels and low diversity.

Natural Recovery and seeded treatments can be compared by how well they function in contrast to the Undisturbed treatment. The Natural Recovery treatment

has a higher erosion risk because of the high amount of bareground. The seeded treatments, while providing greater erosion control, fail to produce a stable, diverse community, probably due to light competition. The function of both seeded and Natural Recovery treatments needs further development before becoming equal to the Undisturbed treatment. The erosion control potential will probably improve with time while the diversity of the seeded treatments may be improved by grazing.

4.0 Conclusions

Natural Recovery treatments produced a community that functioned well in all respects except erosion control was low. Seeded treatments functioned well but had poor species richness. The lack of diversity and richness in seeded treatments may be linked to light competition. Stability of biomass production in the Undisturbed treatment during the 2000 drought provides evidence that diverse communities are more stable. Cattle grazing had structural impacts on seeded and Natural Recovery treatments that will likely improve ecosystem function in seeded treatments but degrade it in Natural Recovery treatments by increasing percent bareground. The "vital ecosystem attributes" framework of Aronson et al. (1993) was useful for organizing many diverse ecosystem measurements.

Table 5-1: Percent cover estimates in five revegetation treatments in mixed grass prairie from 1997 to 2000.

Variable	Soil	Treatment	1997[2]	1998[2]	1999	2000
Live	Chernozemic	P Value	0.001	0.36	0.01*	0.01*
		Standard Error	5.0	5.7	8.1	2.7
		Current	53.0b	49.9b	58.0a	24.1ab
		Diverse	57.6b	48.1b	55.4a	21.6bc
		Simple	51.0b	51.7ab	52.8a	29.7a
		Natural Recovery	36.7c	63.5a	29.9b	18.3c
		Undisturbed	80.0a	57.3ab	62.2a	25.0ab
	Solonetzic	P Value	0.78	0.29	0.01	0.44
		Standard Error	5.4	8.0	6.3	3.4
		Current	41.5ab	45.5a	55.7a	31.9a
		Diverse	46.4ab	63.7a	62.2a	27.4a
		Simple	42.1ab	51.9a	55.7a	30.0a
		Natural Recovery	37.9b	66.9a	30.6b	25.4a
		Undisturbed	53.2a	51.2a	61.1a	29.6a
Litter	Chernozemic	P Value	0	0.19	0.44	0.002
		Standard Error	4.5	6.5	5.1	3.2
		Current	2.2b	52.1a	23.3a	33.0bc
		Diverse	2.9b	35.3b	28.2a	36.5b
		Simple	5.3b	41.4ab	23.8a	26.5c
		Natural Recovery	1.7b	35.3b	21.9a	29.3bc
		Undisturbed	49.2a	43.0ab	18.1a	44.0a
	Solonetzic	P Value	0.02	0.13	0.51	0.17
		Standard Error	4.1	8.0	6.8	5.8
		Current	1.3b	52.3a	22.1a	20.8a
		Diverse	3.1b	42.7ab	16.1a	29.4ab
		Simple	1.6b	42.2ab	13.8a	24.4ab
		Natural Recovery	0.3b	27.8bc	19.3a	20.6b
		Undisturbed	22.9a	19.0c	10.6a	34.4a
Bare ground	Chernozemic	P Value	0.0	0.005	0.04	0.003
		Standard Error	4.1	7.1	8.0	3.2
		Current	47.6a	34.7ab	14.5b	11.6b
		Diverse	42.8a	32.5b	11.8a	11.4b
		Simple	51.5a	44.7ab	19.3b	12.4b
		Natural Recovery	50.0a	48.6a	37.5a	30.7a
		Undisturbed	11.5b	2.5c	12.0b	8.8b
	Solonetzic	P Value	0.001	0.18	0.15	0.12
		Standard Error	3.6	8.1	7.9	9.0
		Current	51.2a	33.5a	14.0ab	8.2b
		Diverse	48.4ab	44.1a	10.b	3.2b
		Simple	40.5b	43.5a	19.8ab	6.8b
		Natural Recovery	55.3a	49.4a	31.2a	29.3a
		Undisturbed	9.6c	6.9b	11.8b	8.5ab

[1] Means with different letters are significantly different within a year

[2]Cover data was collected differently in 1997 and 1998 from 1999 and 2000 so absolute values are not directly comparable

* F-tests recalculated with proc mixed

Table 5-2: Species richness in five revegetation treatments in mixed grass prairie in 1997 to 2000.

Variable	Soil	Treatment	1997	1998	1999	2000
Species Richness	Chernozemic	P Value	0.05	0.04	0.0001*	0.001*
		Standard Error	1.0	1.6	1.4	0.77
		Current	12.3b	9.3b	5.9c	3.6c
		Diverse	16.3a	16.0a	10.8b	5.1bc
		Simple	13.7ab	13.3ab	9.0b	4.1c
		Natural Recovery	11.5b	16.8a	16.3a	7.8a
		Undisturbed	12.5b	13.8ab	9.3b	6.4ab
	Solonetzic	P Value	0.11	0.04	0.02	0.0001
		Standard Error	1.5	1.6	1.3	0.9
		Current	11.7ab	10.3b	8.7b	4.3d
		Diverse	15.3a	18.0a	13.0a	7.0bc
		Simple	12.7ab	14.0ab	11.7ab	5.3cd
		Natural Recovery	10.3b	17.3a	13.0a	8.3b
		Undisturbed	16.0a	17.0a	14.7a	12.7a

[1] Means with different letters are significantly different within a year

*F-test recalculated in proc mixed

Table 5-3: Aboveground biomass components ($g/0.1m^2$) in five revegetation treatments in mixed grass prairie in 1997 to 2000.

Variable	Soil	Treatment	1997	1998	1999	2000
Total	Chernozemic	P Value	-	-	0.001	0.001
		Standard Error	-	-	5.1	6.3
		Current	-	-	54.3a	53.7bc
		Diverse	-	-	54.4a	75.2a
		Simple	-	-	50.3a	65.6ab
		Natural Recovery	-	-	44.7a	45.6c
		Undisturbed	-	-	18.5b	22.6c
	Solonetzic	P Value	-	-	0.003	0.19
		Standard Error	-	-	11.0	12.3
		Current	-	-	68.8a	49.0a
		Diverse	-	-	70.3a	30.8ab
		Simple	-	-	70.3a	39.1ab
		Natural Recovery	-	-	49.0a	44.7ab
		Undisturbed	-	-	11.8b	18.5b
Live	Chernozemic	P Value	0.05	0.01	0.005	0.004*
		Standard Error	2.7	2.9	3.1	2.1
		Current	21.0a	22.0a	25.2a	8.8b
		Diverse	27.1a	23.0a	24.2a	13.7a
		Simple	19.8a	20.5a	21.1a	13.3a
		Natural Recovery	17.1a	18.7a	19.3a	8.8b
		Undisturbed	7.9b	6.3b	11.1b	5.2b
	Solonetzic	P Value	0.22	0.01	0.02	0.02
		Standard Error	5.5	2.5	5.4	2.1
		Current	20.1a	18.9a	30.8a	11.6bc
		Diverse	20.0a	22.4a	31.1a	16.4a
		Simple	8.3a	18.8a	24.8a	15.3ab
		Natural Recovery	20.3a	15.9ab	29.2a	13.1ab
		Undisturbed	3.5a	5.5b	8.7b	8.0c
Litter	Chernozemic	P Value	-	-	0.0001	0.002
		Standard Error	-	-	3.2	6.1
		Current	-	-	29.0a	44.2bc
		Diverse	-	-	30.2a	61.6a
		Simple	-	-	29.2a	54.6ab
		Natural Recovery	-	-	24.7a	37.8c
		Undisturbed	-	-	7.5b	17.5d
	Solonetzic	P Value	-	-	0.005	0.24
		Standard Error	-	-	8.2	11.7
		Current	-	-	38.0ab	37.4a
		Diverse	-	-	39.2a	24.3a
		Simple	-	-	44.6a	35.6a
		Natural Recovery	-	-	19.8bc	35.0a
		Undisturbed	-	-	3.1c	10.7a

[1]Means with different letters are significantly different within a year

- Data not collected

* F-test recalculated in proc mixed

Table 5-4: Soil nitrate (NO_3), ammonium (NH_4) and total mineral nitrogen (MinN) in parts per million (ug/g) for the upper 10 cm of soil in five revegetation treatments in mixed grass prairie in 1996 to 2000.

Variable	Soil	Treatment	1996	1997	1998	1999	2000
Nitrate	Chernozemic	P Value	0.36	0.01	0.38	0.51*	0.55*
		Standard Error	2.4	1.3	0.9	1.1	4.7
		Current	11.5a	4.2ab	5.3a	2.5a	7.2a
		Diverse	17.6a	7.4a	5.5a	3.2a	9.2a
		Simple	12.8a	8.8a	5.1a	2.7a	8.7a
		Natural Recovery	17.3a	6.7ab	5.9a	3.9a	12.5a
		Undisturbed	11.3a	1.2b	3.4a	2.4a	5.9a
	Solonetzic	P Value	0.28	0.04	0.35	0.02	0.05
		Standard Error	3.9	0.9	0.9	1.7	1.6
		Current	20.9a	3.7ab	3.4a	1.7b	3.4b
		Diverse	14.8a	5.8ab	4.1a	3.5b	3.6b
		Simple	18.7a	4.7ab	2.7a	2.2b	4.2b
		Natural Recovery	18.9a	7.5a	5.3a	8.7a	9.4a
		Undisturbed	7.6a	0.9b	2.3a	2.2b	3.7b
Ammonium	Chernozemic	P Value	0.23	0.55	0.43	0.67*	0.05*
		Standard Error	0.8	1.0	1.2	0.31	0.7
		Current	5.9a	3.9a	6.0a	1.8a	2.4b
		Diverse	5.5a	6.1a	6.3a	2.0a	3.5ab
		Simple	3.8a	4.8a	4.1a	1.9a	3.7ab
		Natural Recovery	5.1a	4.5a	3.6a	1.6a	3.2b
		Undisturbed	3.4a	3.9a	4.7a	2.0a	4.5a
	Solonetzic	P Value	0.82	0.45	0.30	0.51	0.77
		Standard Error	0.8	0.7	0.5	0.34	0.72
		Current	4.1a	3.9a	4.5a	1.6a	4.3a
		Diverse	5.2a	4.0a	5.3a	1.8a	3.6a
		Simple	4.0a	4.1a	3.8a	1.5a	3.3a
		Natural Recovery	3.9a	5.5a	4.5a	1.9a	3.6a
		Undisturbed	4.1a	3.4a	5.2a	1.4a	3.7a
Mineral Nitrogen	Chernozemic	P Value	0.07	0.04	0.56	0.65	0.66*
		Standard Error	2.1	2.0	1.8	1.1	4.7
		Current	17.5a	8.1a	11.4a	4.3a	9.6a
		Diverse	23.1a	13.5a	11.8a	5.2a	12.7a
		Simple	16.7a	13.5a	9.2a	4.6a	12.4a
		Natural Recovery	22.4a	11.2a	9.5a	5.5a	15.8a
		Undisturbed	14.8a	5.1a	8.0a	4.0a	10.4a
	Solonetzic	P Value	0.31	0.08	0.40	0.02	0.11
		Standard Error	4.0	1.4	1.3	1.8	2.4
		Current	25.0a	7.6ab	7.9a	3.3b	7.7b
		Diverse	20.0a	9.7ab	9.4a	5.3b	7.2b
		Simple	22.8a	8.8ab	9.9a	3.7b	7.4b
		Natural Recovery	22.8a	12.9a	6.4a	10.7a	13.0a
		Undisturbed	11.7a	4.0b	7.5a	3.6b	7.2b

[1] Means with different letters are significantly different within a year

*F-test recalculated using proc mixed

Table 5-5: Available percent light in five revegetation treatments in mixed grass prairie in 1999 and 2000.

Soil	Treatment	1999	2000
Chernozemic	P Value	0.05	0.0006
	Standard Error	0.07	0.04
	Current	0.46abc[1]	0.66b
	Diverse	0.37c	0.64b
	Simple	0.44bc	0.61b
	Natural Recovery	0.54ab	0.70b
	Undisturbed	0.63a	0.87a
Solonetzic	P Value	0.004	0.0017
	Standard Error	0.07	0.06
	Current	0.24b	0.48b
	Diverse	0.15b	0.46b
	Simple	0.22b	0.38b
	Natural Recovery	0.41a	0.70a
	Undisturbed	0.50a	0.69a

[1] Means with different letters are significantly different within a year

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Chapter VI Cattle Grazing Selection and Mixed Grass Prairie Wellsite Revegetation

1.0 Introduction

Restoration activities often focus on the landscape scale (Naveh 1994). Rather than focusing on reestablishing communities or ecosystem function at a particular place, restoration is a process of transforming entire landscapes. Landscape restoration may occur through the cumulative effects of many small restoration activities, but the execution of each of these smaller projects occurs within a landscape perspective (Palik et al. 2000). We must ask how small scale restorations functions as landscape elements (patch, corridor, network) within the entire landscape (Forman 1997). For example, we must ask whether the restored community or ecosystem is compatible with the surrounding vegetation, animal activities (Archer and Pyke 1991) and land uses.

Revegetation of petroleum development on native MGP provides an example of small scale restoration projects that should be considered within a landscape context. Pipelines, wellsites, access roads and associated infrastructure are all common disturbances on remnant MGP. Over 30,000 active and 10,000 abandoned wells were in the dry mixed grass subregion of Alberta in 1991 (Kerr et al. 1993). Well site construction usually involves the destruction of vegetation and stockpiling of topsoil and subsoil on an area 100 x 100 m.

Current petroleum developments (wellsites, pipelines) on MGP are currently typically seeded to native species cultivars. Long term impacts of these disturbances depends on how revegetated areas function within a landscape context. In MGP, typically used for cattle grazing, if seeded species cannot withstand grazing pressure

further degradation may occur. Alternately, if a site is revegetated with species incompatible with the grazing on adjacent land, the long term impact may be permanent loss of the grazing resource and even further degradation of the range if incompatible seeded species invade the undisturbed prairie.

An example of this problem is the use of *Agropyron pectiniforme* R. & S. (crested wheatgrass) to revegetate wellsites in the MGP. Crested wheatgrass is best utilized in early spring when it is most palatable to cattle (Smoliak 1968), while grazing on native range should be deferred as late as possible. Having both types of forage resources within one field presents a management problem at the landscape scale. Either the utilization of the crested wheatgrass is forgone or condition of the native range is sacrificed by an early cattle entry date.

Current revegetation of wellsites in the MGP typically utilizes a mixture of *Agropyron dasystachyum* (Hook) Scibn. (northern wheatgrass), *Agropyron trachycaulum* (Link) Malte (slender wheatgrass) and *Agropyron smithii* Rydb (western wheatgrasss). These species represent a significant improvement over the use of *Agropyron pectiniforme,* which was common only 10 years ago (Gerling et al. 1996). It has yet to be demonstrated, however, that areas seeded to these wheatgrass cultivars will withstand grazing pressure or be compatible with undisturbed MGP.

This chapter will focus on the compatibility of revegetated wellsites as a grazing resource within the landscape context. To guarantee that revegetated wellsites in MGP will be utilized by cattle, it is necessary to seed species that are both palatable and preferred by cattle at the time of year when they are typically on the range. Likewise, disturbances should not be revegetated in a way that causes them to

be preferentially used and thus overgrazed. A revegetated disturbance within native prairie represents a patch (landscape element) that cattle may graze or avoid.

Animal utilization of patches can be determined by utilization cages (e.g. Smoliak and Slen 1974) or by observation (e.g. Sheehy and Vavra 1996, Biondini et al. 1999). Other methods include tracking utilization of individual plants (e.g. Willms et al. 1980, Willms and Rode 1998) or measuring stubble height (e.g. Osko et al. 1993). Since the objective of this study was to measure utilization at the patch scale, utilization cages were initially chosen as the main method of assessing patch use. In a second year of data, behavioral observations were added because grazing cage data showed considerable variability.

Most studies of patch grazing selection involve unreplicated observation of cattle use of various patch types and then developing utilization indices by comparing utilized with available habitat. These data are typically assessed using a chi square test (e.g. Pinchak et al. 1991) or discriminant function analysis (e.g. Sheehy and Vavra 1996). A more experimental approach is to constrain an animal's landscape position (fence within a small area) and present them with a choice of patches within which to graze. For example, Smoliak and Slen (1974) tested cattle preference between native range, *Agropyron pectiniforme* and *Elymus junceus* Fisch. (russian wildrye) in MGP by placing cattle within paddocks that contained all three patch types. Willms et al. (1980) tested deer and cattle preference for burned or unburned treatments by enclosing animals in areas < 1 ha with both patch types.

This study investigated patch selection by cattle of three seeded, Natural Recovery and Undisturbed MGP treatments within a 1 ha area. The general objective

was to determine whether cattle would preferentially graze or otherwise spend time in one or more specific revegetation treatments.

2.0 Methods

2.1 Site description

The study took place in four sites of native prairie in southeastern Alberta, Canada from May 1999 to August 2000 (Figure A-1). Two sites were located 300 m apart, approximately 20 km northwest of Medicine Hat on loam textured Orthic Brown Chernozemic soils. The remaining two sites were 3 km apart, 15 km northeast of Bow Island on sandy loam textured Orthic Brown Chernozemic soils. Natural vegetation in this area is referred to as Dry Mixed Grass Prairie (Strong and Leggat 1992) or MGP (Coupland 1961). Undisturbed prairie at each of the sites was dominated by *Stipa comata* Trin & Rupr. (needle and thread) and *Bouteloua gracilis* (HBK) Lag. (blue grama grass). Taxonomy follows Moss (1983). The climate is semiarid and the majority of growing season precipitation falls in June. May to September precipitation was slightly below long-term averages in 1996 to 1998 and moderately above long-term averages in 1999 (Table A-1). In 2000 precipitation was approximately half the long-term averages.

Each of the research sites was a drilled and abandoned wellsite located inside a field of native prairie used for grazing. Cattle have grazed each of the fields for over thirty years. In 1999, the range condition of the sites was estimated to vary from good to excellent (Wroe et al. 1988).

Wellsites were established in either fall 1995 or spring 1996 and abandoned in spring 1996. Each company contributing wellsites was asked to reclaim them using

normal practices up to the point of being seeded. Each wellsite had the topsoil and subsoil separately removed and stockpiled. After industrial activities were finished, topsoil and subsoil were replaced and a seedbed prepared. Each wellsite and an adjacent strip of undisturbed native prairie were fenced to exclude grazing for the first three years of the study.

2.2 Treatments and experimental design

Three seeded and one non-seeded (Natural Recovery) treatments were compared to the Undisturbed treatment in this study. The seeded treatments consisted of a three species, wheatgrass dominated mix (Current), a five species mix (Simple) and a 21 species mix (Diverse) (Table A-2). All treatments were investigated with and without cattle grazing.

Each of the wellsites was treated as a block. The 3 seeded and Natural Recovery treatments were placed in the same location within each block because of the influence of predominant wind direction on natural revegetation (Table A-2). The position of the Undisturbed treatment however, varied with each block. Exclosures allowed comparison of Ungrazed and Grazed areas within each revegetation treatment. The experiment was treated as a strip plot design.

2.3 Treatment application

Revegetation treatments were implemented in spring 1996 by Hammermeister (2001). Seeding rates (kg/ha) were calculated based on desired number of pure live seeds (PLS) per unit area (300 PLS/m^2). Because purity and viability data were not available for forbs, they were estimated as 66%. Chick starter was used as a carrier to ensure even seed distribution and improve seed flow. The total seeding rate was

applied with two perpendicular passes of a calibrated Truax native seed drill. *Koeleria macrantha* (Ledeb.) J.A. Schultes f. (june grass) and forbs were broadcast with chick starter following seeding but before straw crimping. Although the seeder was calibrated, seeding rate varied slightly due to limited accuracy of openers (10%).

During summer 1999 grazing exclosures were built in the center of each site enclosing part of each revegetation treatment. The unfenced portion of each treatment was grazed in June and July 1999. Grazing treatments were applied by moving cattle from the adjoining field into the fenced wellsite where the animals were free to graze any of the treatments outside the exclosure. Water was provided in troughs placed directly beside the gate of each wellsite. Cattle remained in the wellsite until visually estimated utilization approached 30 to 50% on each treatment. A confounding factor to the experimental design is that two sites were mowed two weeks prior to grazing in 1999. Because of this complication, stocking rates of the Grazing treatment were reduced for these two sites (Table A-3). In summer 2000 mowed and unmowed sites were visually similar and all blocks were included in the analysis. Because of drought conditions in 2000, stocking rates were dropped.

2.4 Data collection

In summer 1999 when the initial grazing treatments were applied, three randomly located 0.1 m^2 quadrats were clipped in both Grazed and Ungrazed portions of each revegetation treatment to determine utilization. All above ground biomass was collected one to two weeks after grazing, dried in an oven at 60 °C and weighed.

In summer 2000, three 1 x 2 m grazing cages were randomly placed in each Grazed subplot. After application of the grazing treatment, a 0.25 m^2 quadrat was clipped in the center of each grazing cage. Three randomly placed 0.25 m^2 quadrats

were clipped outside the cages in each subplot. Above ground biomass was coarsely sorted into standing litter, ground litter and live vegetation. Because of considerable variation between subamples, an additional three quadrats were clipped in and outside the cages in each subplot one to two weeks after the initial clipping. July 2000 was exceptionally dry and little growth took place between clippings. The exception was at one site where *Kochia scoparia* (L.) Schrad. (kochia weed) grew considerably between clippings. This new growth was omitted from samples collected because it was not part of available forage at the time of grazing. Adding the second round of clipping data did not change results considerably.

Behavioral observations were made during application of grazing treatments in June to July 2000. A semi-structured sampling scheme was used to make observations. Three to four sampling periods (usually on consecutive days) were established where observations on cattle were made at one hour intervals. Length of sampling period varied between one and ten hours. Each observation consisted of recording the number of cattle either grazing (biting, searching or moving between feeding stations) or resting (sleeping, ruminating) in each treatment. If cattle behavior was not classifiable into these categories (e.g. drinking from water trough), was influenced by the observer or were conducted on the buffer edges outside the treatment areas, cattle activity was classified as "other". Observations for both mature animals and calves were collected. However, only data for mature animals was analyzed because calf behavior generally reflected that of mature animals.

2.5 Data analysis

2.5.1 Utilization data

Utilization from grazing cages is typically determined with the paired subplot method where paired quadrats inside and outside the grazing cage are directly compared. Because treatments were visually homogeneous, quadrats in grazed areas and cages were not paired but placed randomly in 2000. In 1999, the Ungrazed subplots were treated as one large grazing cage. Clipping data from all quadrats from within a plot were averaged to give one value for Grazed and Ungrazed components of the plot. If data points were missing, then the number of data points available for any one plot were averaged to calculate the value for that plot. In 2000, the actual number of quadrats per plot varied between 4 and 6.

Utilization can be calculated as an absolute difference (cage – grazed) or as a percent (absolute difference ÷ cage). Percent utilization emphasizes impacts on the plant community while absolute utilization emphasizes the grazing animal (Bork and Werner 1999). Both calculations were made and analyzed in this study to investigate different facets of the grazing system. The calculation of utilization hinges on the definition of available forage. Usually ANPP (current year's growth) is assumed to represent available forage (e.g. Bork and Werner 1999). However, other sward components such as litter can be consumed, destroyed by trampling or moved from one sward component to another (standing to ground litter) by cattle. To investigate these other dynamics, utilization was calculated for live vegetation, standing litter and ground litter and select combinations of these components.

Data were analyzed as a randomized block design using analysis of variance (ANOVA). Treatment means were compared using the least significant difference

(lsd) at alpha equal to 0.05. Minitab software was used to analyze the data using proc glm. Given the small sample size, tests were not made for normality or homogeneity of variance. ANOVA is generally regarded as robust to violations of both of these assumptions (Day and Quinn 1989). The third assumption of ANOVA, independence of samples, is also violated in this study by the lack of randomization. The fixed location of treatments in this study increases the risk of a Type I error however many precedents exist to justify this sort of analysis in field studies and choice of analysis should reflect the judgement and objectives of the researcher rather than inflexible statistical rules of thumb (Stewart-Oaten 1995).

2.5.2 Behavioral data

All observations were pooled for each block to calculate a total number of observations of cattle grazing or resting in each treatment. " Other " observations were discarded for the purpose of analysis since they were due to factors that varied between blocks. To use parametric tests, it was necessary to standardize the observations since the number of observations varied between blocks. To accomplish this, the number of observations of grazing and resting per treatment was divided by the total number observations made at that block. This calculated value represents the percent time spent either resting or grazing in each treatment while the animal was within the scope of the study (not outside plot boundaries or conducting an unclassifiable activity).

Behavior data were analyzed using analysis of variance (ANOVA) treating the data as a randomized block design as described above for clipping data using SAS software (SAS Institute Inc, 2000). The relative size of each treatment varied within blocks, so to test if cattle were selecting areas required incorporation of the amount of

each treatment available. The area of each plot was determined and divided by the area of all plots within a block to determine the percent area of each treatment. The percent area was used in an analysis of covariance (ANCOVA) on percent time resting and grazing in each treatment.

The location of the water trough also varied in each block and so treatments were ranked from 1 to 5 on their proximity to water. Proximity to the water tank was used as a covariate in an ANCOVA on percent time spent resting and grazing in each treatment. If a covariate was significant in an ANCOVA then the covariate by treatment interaction was tested to detect heterogeneity slopes.

3.0 Results and Discussion

3.1 General results

Utilization did not show significant treatment differences in either year for any sward component (Table 6-1). Mean percent utilization on the live sward component (NPP) ranged between 23 and 42% (Table 6-2). Mean absolute utilization of the live sward component ranged from 16 to 40 g/m^2. Mean absolute utilization values for other sward components varied widely from large negative (-103.44 g/m^2) to large positive values (64.84 g/m^2). Mean percent utilization for other sward components ranged between -53% to 32%. An unexpected large number of sward components had negative means particularly in the Current and Diverse treatments.

Time grazing and resting both showed non-significant treatment and block effects in the initial model (Table 6-3). Mean time spent grazing in each treatment varied between 8% in the Undisturbed treatment and 17.2% in the Simple without

inclusion of any covariates (Table 6-4). Mean time spent resting varied between 2.2% in the Natural Recovery treatment to 14.2% in the Undisturbed.

Area of treatment was non-significant when included as a covariate for time grazing and near significant for time resting. Inclusion of area of treatment as a covariate did not yield significant treatment or block effects for either time grazing or resting. Distance to trough was non-significant when included as a covariate for percent time resting but was highly significant when included with time grazing. Tests for heterogeneous slopes of the covariate (trough by treatment interaction) were non-significant. Inclusion of distance to trough as a covariate resulted in near significance for treatment effect. Mean comparison tests of treatments adjusted for distance to trough indicates that a significantly larger amount of time was spent grazing in the Simple treatment over the Undisturbed (Table 6-4).

3.2 Utilization

Grazing cage data showed no significant difference in cattle use among treatments. This result holds for both animal grazing impact on the plant community (percent utilization) and the amount of forage harvested by the animal (absolute utilization). Three alternatives are available for explaining non-significance of the data. Firstly, the results could be taken at face value. That is, in landscape with patches of native prairie, seeded and Natural Recovery wellsites, cattle will graze each of these patches. This interpretation confirms the casual observation that cattle had an impact on each of the treatments available to them.

Secondly, the lack of significance could reflect the inherent variability in the clipping data (i.e. lack of precision). Even with a rigorous sampling design and a relatively homogeneous sward, high variability can be found in biomass clipping

data. For example, Kowalenko and Romo (1998) conducted simple defoliation experiments on a homogenous mixed prairie sward with ten replicates and still found a large amount of variability in the data. In this study, the large number of quadrats (6) used in sampling each plot should have been sufficient to account for subsampling error. Secondly, since data did not change substantially from the first clipping (first three) to the second, higher confidence can be placed in the precision of the data.

A final explanation for the lack of significance could be the lack of precision between replicates. That is, important factors varied between blocks that were not controlled and likely increased variability in the data. For example, different class and breeds of stock were used for various blocks. The past grazing history of cattle used at each block varied in unknown ways. Blocks were slightly different sizes and shapes and were grazed at slightly different times over a one month period. It is possible that these confounding factors have masked important differences in cattle preference.

3.3 Behavioral data

Behavior data provide evidence that some selection was occurring among treatments. Mean comparison tests indicate that limited grazing selection is occurring, with the Simple treatment selected above the Undisturbed. It is unclear why the Simple treatment would be selected over other seeded treatments. The Simple treatment has a higher proportion of *Stipa comata*, a preferred species, than the other seeded treatments however this species makes up only a small proportion of the sward (<5% cover) (Chapter IV). The Simple treatment also lacks *Agropyron trachycaulum* that is present in the other two seeded treatments. This species is a large bunchgrass that is wolfy and probably avoided by cattle.

Cattle did not distinguish among treatments for time spent resting. Near significance of the area of treatment covariate supports this interpretation. If size of the treatment affects the likelihood of cattle resting then cattle are likely not selecting among treatments.

3.4 Synthesis of data and management implications

Cattle grazing selection occurs across a hierarchy of scales. Cattle must choose a landscape position, a patch to graze in, a feeding station and plant parts to bite. Wellsite location represents a landscape position and the seeded wellsite or adjacent native prairie represent patches at such a location that an animal may choose to graze in. Choice of feeding station within these patches or plant parts is inconsequential if our objective is simply to guarantee that the area will be grazed. Fixed factors such as topography, distance to water (Pinchak et al. 1991) and manageable factors such as habituation, salt and oilers influence landscape position of animals. Assuming animals can be managed to be in the vicinity of wellsites, the major component of cattle grazing selection then is the patch scale.

Utilization data indicates that cattle either do not selectively graze any of the treatments but behavioral data indicates that they spend more time in the Simple treatment than the Undisturbed. These two data sets are not necessarily contradictory since time spent grazing in a particular treatment does not need to reflect the amount of forage removed. Bite rates, bite size and searching time between feeding stations can all influence the amount of forage harvested in relation to the amount of time spent in a particular patch. Whether some selection is occurring between patches or none at all it would seem a fair generalization that all revegetation treatments were

utilized by cattle for grazing. Cattle did not consistently avoid any treatment (except the Natural Recovery for resting), nor was any treatment consistently overused.

Assuming that the patch selection of cattle confined to a small area (our experimental system) will be similar when unconfined (actual landscape), this study has important implications for wellsite revegetation within a landscape context. Revegetation practitioners are often concerned that grazing will impede establishment on disturbed areas (Osko et al. 1993). If cattle selectively graze a revegetation treatment there is the danger that such a treatment will be excessively grazed and plant establishment will fail. This study indicates that no treatment was selectively grazed. For revegetation practitioners this means that grazing should be a safe management practice four years after establishment as long as treatments can withstand moderate grazing pressure.

Range managers in MGP need to know whether revegetated wellsites will be eventually used by livestock (Wood et al. 1995) and thus, become a functioning component of the landscape. This study provides evidence that wellsites seeded to predominantly native wheatgrass cultivars can be grazed by cattle during the period when adjacent native range is grazed. Neither native range nor disturbances seeded to wheatgrasses will be used exclusively. Even disturbed areas not seeded will be used and so provide grazing opportunities to livestock.

It is important to note that the results of this study are based on the assumption that the experimental design of this study accurately replicated the grazing behavior of cattle on the open range. There is a strong possibility that cattle grazing behavior may be very different under actual management situations. The predictions based on

this research represent the best information available at this time. Results are presented with the caution that further investigation is required.

4.0 Conclusions

Little selection was detected in either cattle resting or grazing activities between seeded, Natural Recovery and Undisturbed treatments. Patches revegetated by seeding wheatgrass cultivars or allowing Natural Recovery will be utilized by cattle but not at the exclusion of Undisturbed range. The seeding treatments examined should function well within the mixed prairie landscape by providing both a compatible grazing resource and vegetative cover which will not be overgrazed. Natural Recovery treatments function well within a mixed prairie landscape as patches that will be grazed.

Table 6-1: P-values of utilization of five revegetation treatments in mixed grass prairie.

Sward component	Percent Utilization		Absolute Utilization	
	Block Effect	Treatment Effect	Block Effect	Treatment Effect
1999				
Total biomass	0.2415	0.2526	0.1433	0.5819
2000				
Total biomass	0.2777	0.1640	0.3706	0.5713
Total litter	0.1332	0.2954	0.2009	0.4658
Standing litter	0.6196	0.3608	0.8670	0.4874
Ground litter	0.1548	0.1563	0.2811	0.2267
Live	0.7832	0.8999	0.2541	0.6637
Live + standing litter	0.9157	0.2273	0.9873	0.5619

Table 6-2: Mean absolute (g/m^2) and percent utilization values on five revegetation treatments in mixed grass prairie.

Variable	Sward component	Standard Error	Current	Diverse	Natural Recovery	Simple	Undisturbed
1999							
Percent Utilization	Total biomass	11.68	35.25	14.50	35.80	9.25	42.00
Absolute Utilization	Total biomass	16.66	54.31	30.01	38.04	15.00	31.27
2000							
Percent	Total biomass	13.21	-27.47	-2.72	6.85	-7.92	22.69
Utilization	Total litter	15.15	-35.36	8.66	3.11	-13.77	11.25
	Standing litter	21.90	-12.98	3.87	-42.96	13.00	14.86
	Ground litter	19.96	-53.09	-9.04	16.87	30.24	7.00
	Live	14.06	37.62	33.72	22.83	33.13	42.03
	Live + standing litter	12.57	3.11	6.97	-10.83	18.46	31.82
Absolute	Total biomass	64.56	-82.72	-22.72	33.24	0.84	64.84
Utilization	Total litter	55.95	-103.44	-56.32	13.92	0.92	26.56
	Standing litter	28.85	-13.6	7.84	-41.76	31.08	9.56
	Ground litter	44.09	-90.92	-39.13	45.36	-37.36	14.64
	Live	11.40	24.27	39.67	15.98	23.63	20.76
	Live + standing litter	30.95	15.12	21.24	-22.44	49.26	37.00

Table 6-3: ANOVA and ANCOVA results of cattle selection of five revegetation treatments in mixed grass prairie.

Variable	Covariate	Covariate Significance	Block Significance	Treatment Significance
Percent time grazing	None	NA	0.82	0.47
	Area of treatment	0.98	0.84	0.54
	Distance to trough	0.002	0.57	0.19
	Trough by treatment interaction	0.74 (0.03)*	0.68	0.89
Percent time resting	None	NA	0.94	0.42
	Area of treatment	0.12	0.93	0.39
	Distance to trough	0.26	0.94	0.44

*P-value of the trough term

Table 6-4: Mean percent time spent grazing and resting by cattle in five revegetation treatments in mixed grass prairie.

Variable	Covariate	Standard Error	Current	Diverse	Natural Recovery	Simple	Undisturbed
Percent time grazing	None	5.0	12.7	10.0	12.2	17.2	8.0
	Area of treatment	5.0	12.7	10.0	12.2	17.2	7.8
	Distance to trough	3.2	10.5	12.0	11.4	17.2	8.7
	Trough by treatment interaction	3.9	10.8	12.0	11.1	17.2	10.4
Percent time resting	None	7.7	14.2	4.7	2.2	4.9	14.1
	Area of treatment	7.4	16.8	8.2	1.6	5.3	8.0
	Distance to trough	7.7	13.0	6.1	1.6	4.9	14.6

5.0 Literature Cited

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Chapter VII Recommendations for Mixed Grass Prairie Wellsite Revegetation

1.0 Introduction

This thesis took a hierarchical approach to study mixed grass prairie (MGP) wellsite revegetation. Many definitions of restoration can be organized along a hierarchy of scales, complex ecosystems can be understood as a hierarchy of scales and there is a concordance of scales between ecosystems and definitions of restoration. The intent of this approach was to attain an understanding of the process of wellsite revegetation and therefore, produce good management recommendations.

At this point it is necessary to distinguish between the processes and study of restoration (restoration ecology) and management with a restoration goal (ecological restoration). Ecological restoration is used to refer to the actual management activities that attempt to reestablish plant communities, ecosystem function, etc. While ecological restoration may be a process, it always has a goal or *telos* that is usually scale specific. Ecological restoration activities attempt to achieve a specific goal at a specific scale (e.g. establish a *Stipa-Bouteloua* community on a 1 ha wellsite) (Allen and Hoekstra 1987). Restoration ecology is the study of the processes that occur during restoration. These processes necessarily occur at all scales. That is, one cannot attempt to restore a *Stipa-Bouteloua* community without impacting ecosystem function, population patterns and landscape function. Management goals may not exist at all of these scales. However to understand what is happening during the restoration process, we must understand what is happening across all scales.

2.0 Linking scales

Studying the restoration process across a range of scales should allow a more complete understanding it. The key to understanding the restoration process across a range of scales however is to link processes across scales. Two mechanisms are possible for the linking of scales: larger scales provide the context in which smaller scales behave (constrain) and smaller scales provide the mechanisms by which to explain the behavior of higher scales (signal) (Allen and Hoekstra 1992, Allen and Starr 1982).

MGP wellsite revegetation in this study can be described at five scales: population, community, ecosystem, landscape and regulatory (social). By considering in turn how each scale is linked to those above and below it, we can create a working model of the wellsite revegetation process.

3.0 A hierarchical model of mixed grass prairie revegetation

The largest scale in the wellsite revegetation process is the regulatory scale. This social scale provides the context for all other scales. The regulatory structure in Alberta dictates what revegetation practices will be enacted to affect the revegetation process at the landscape, ecosystem, community and population scales. The current regulatory framework emphasizes achieving a high amount of ground cover in as short a time as possible, so revegetation practices consist of planting aggressive wheatgrass cultivars (AEP 1995). If Natural Recovery and early cattle grazing were to be adopted as management strategies, changes would have to occur at the regulatory scale. There is only weak feedback from lower levels to the regulatory scale. However, if, for example, Natural Recovery turns out to be a superior revegetation technique, this superior performance may influence changes in the regulatory framework.

The highest empirical scale in wellsite revegetation is the landscape scale. This scale was investigated with regards to cattle grazing. Whether cattle will choose to graze on revegetated wellsites has profound impact on the wellsite revegetation process. This research has presented evidence that cattle will graze wellsites revegetated with wheatgrass cultivars or Natural Recovery treatments. If we accept preliminary evidence from this research this means that if wellsites are not fenced they will be grazed but not likely overgrazed. The choice to exclude grazing therefore becomes an important management decision to determine the landscape context of the wellsite. Cattle grazing selection constrains lower levels of ecosystem function, community succession and populations by determining whether cattle grazing will be present. Communities and populations ultimately provide part of the mechanism by which cattle grazing selection occurs by determining the species that will be present.

Ecosystem function is the second largest empirical scale investigated in this study. Ecosystem function provides the context for community development. Nutrient cycling, energy flow, erosion control and diversity are all ecosystem attributes and functions that provide the context for successional trends. Two plant communities may have similar species composition but if they differ in nutrient cycling or erosion control, different successional trends may occur. This study determined that seeded treatments functioned well but diversity was low. Natural Recovery treatments functioned well except erosion risk may be increased. Cattle grazing has the potential to improve functioning in seeded treatments by increasing available light but will likely degrade function in the Natural Recovery treatment by increasing bareground.

The community scale was investigated by examining successional trends and cover of major species. Community succession provides one of the mechanisms to explain ecosystem function while at the same time provides the context for population changes. This study found that seeded treatments will become dominated by wheatgrass cultivars within the first five years of establishment and will may remain that way indefinitely. This community trend is the mechanism by which the function of seeded treatments has adequate erosion control but poor diversity. Ecosystem function also provides context for community change in the seeded treatments. Grazing increases available light and so may allow estblishment of non-wheatgrass species.

Natural Recovery treatments either progress towards a composition similar to the Undisturbed treatment or remain dominated by early seral grasses and weeds. Succession in the Natural Recovery treatment is constrained by the ecosystem function scale when grazed since increased erosion risk may impede community development.

The smallest scale investigated was that of population. Both plant demographics and seed bank composition were investigated at this scale. Ultimately the birth and death of plants provides the mechanism by which community and ecosystem changes occur. This study found that the seed bank probably was an important source of propagules for the Natural Recovery treatment (weeds, bunch grasses) but a less important source of plant recruitment in seeded treatments. Plant demographics indicated that plants are reproducing faster in the Natural Recovery than in seeded treatments.

4.0 Management Recommendations

By linking ecosystems across a hierarchy of scales, it is possible to construct a simple conceptual model of the wellsite revegetation process (Figure 7-1). The results of

different management actions can then be predicted. Seeding wellsites with seed mixes that include wheatgrass cultivars will result in good erosion control but poor community development and low diversity. The mechanism of this problem is probably competition for light. It may be possible to alleviate this problem through cattle grazing.

Natural Recovery treatments will result in higher erosion risk but otherwise function well. Natural Recovery communities will likely establish predisturbance plant communities if given enough time. Rapid population expansion is a possible mechanism of this community trend. Cattle grazing will negatively impact Natural Recovery treatments by increasing bareground and the number of weed seeds in the seed bank.

Based on the above conceptual model the following management recommendations can be made for the revegetation of MGP wellsites.

1. For complete erosion control, native wheatgrass cultivars provide an effective, rapid solution for mixed prairie wellsite revegetation.
2. Inclusion of native wheatgrass cultivars even in small percentages of a seed mix will result in communities that are wheatgrass dominated, have low species diversity and will not likely return to predisturbance conditions.
3. Cattle grazing on mixed prairie wellsites revegetated with wheatgrass cultivars four years after seeding can be used to curb aggressive wheatgrasses, and potentially increase establishment of less dominant species.
4. Natural Recovery is an effective long-term strategy for reestablishing the predisturbance plant community although risk for erosion is higher than for seeded treatments.

5. Cattle grazing is not an appropriate management strategy for Natural Recovery wellsites as it will impede community development and ecosystem function.

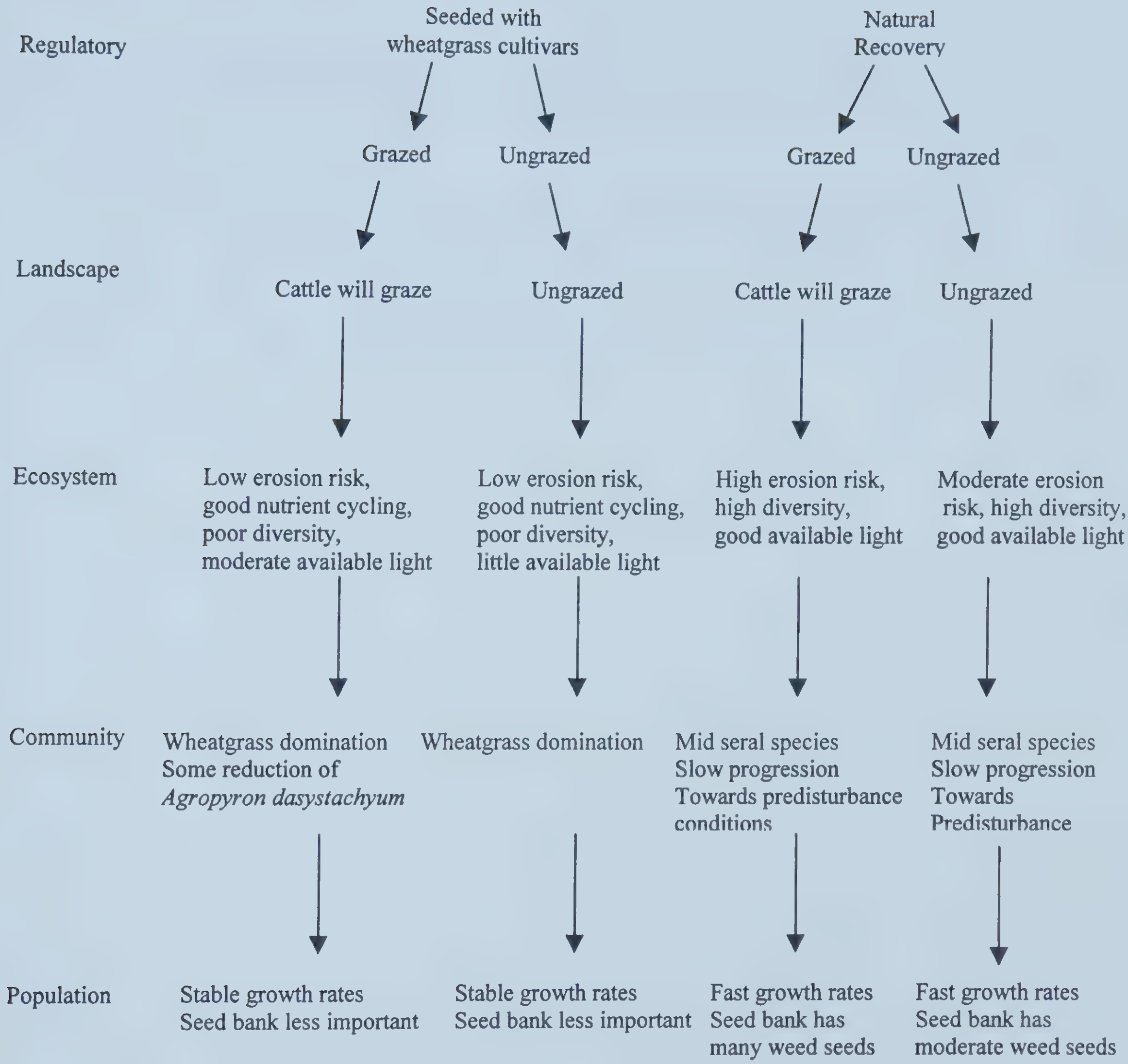

Figure 7-1: Hierarchical model of wellsite revegetation in mixed grass prairie.

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APPENDIX I – SITE AND TREATMENT INFORMATION

Table A-1:Total precipitation (mm) and air temperature at locations near the research sites (May to September).

	1996	1997	1998	1999	2000	LTA[1]
Precipitation						
Brooks	192	141	176	312	101	218
Bow Island	174	133	195	208	114	211
Medicine Hat	172	141	279	159	106	207
Temperature						
Brooks	15.2	16.3	17.1	14.1	15.3	15.0
Bow Island	16.1	17.1	18.3	14.5	16.0	16.4
Medicine Hat	15.0	16.7	18.1	14.5	16.9	16.3

[1]LTA - Long term average (Brooks, 1961-1988; Bow Island and Medicine Hat, 1961-1997)

Table A-2: Description of treatments and plot location.

Treatment	Location	Seeded Species
Natural Recovery	Southwest	Not seeded
Current	Southeast	*Agropyron smithii, Agropyron dasystachyum, Agropyron trachycaulum, Stipa viridula*
Simple	Northeast	*Stipa comata, Bouteloua gracilis, Koeleria macrantha, Agropyron smithii, Agropyron dasystachyum*
Diverse	Northwest	*Bouteloua gracilis, Stipa comata, Agropyron smithii, Agropyron dasystachyum, Agropyron trachycaulum, Koeleria macrantha, Stipa viridula, Oryzopsis hymenoides, Elymus canadensis, Vicia americana, Ratibida columnifera, Achillea millefolium, Gutierrezia sarothrae, Petalostemon purpureum, Aster ericoides, Astragalus striatus, Gaillardia aristata, Petalostemon candidum, Hedysarum sulphurescens, Thermopsis rhombifolia*
Undisturbed	Adjacent to wellsite disturbance	Not seeded or disturbed

Table A-3: Grazing treatments applied in summer 1999.

Site	Grazing Period	Type of Stock	Stocking Rate (AUM/ha)
Intensity West	June 25 (1630 h) to June 28 (0830 h)	3 cow-calf pairs, 1 steer, 4 dry cows	0.78
Intensity East	June 29 (1000 h) to July 1 (1600 h)	3 cow-calf pairs, 1 steer, 4 dry cows	0.66
Sceptre	July 16 (1500 h) to July 18 (1000 h)	20 cow-calf pairs, 1 bull	1.90
Startech	July 1 (1730 h) to July 3 (1400 h)	20 heifers, 1 bull	1.55

Table A-4: Grazing treatments applied in summer 2000.

Site	Grazing Period	Type of Stock	Stocking Rate (AUM/ha)
Intensity West	July 4 (2000 h) to July 9 (1300 h)	5 cow-calf pairs, 1 steer	0.97
Intensity East	July 9 (1300 h) to July 13 (1500 h)	5 cow-calf pairs, 1 steer	0.86
Sceptre	June 26 (1200 h) to June 28 (1630 h)	12 cow-calf pairs, 1 bull	1.54
Startech	June 22 (0930 h) to July 25 (1730 h)	13 steers	1.57

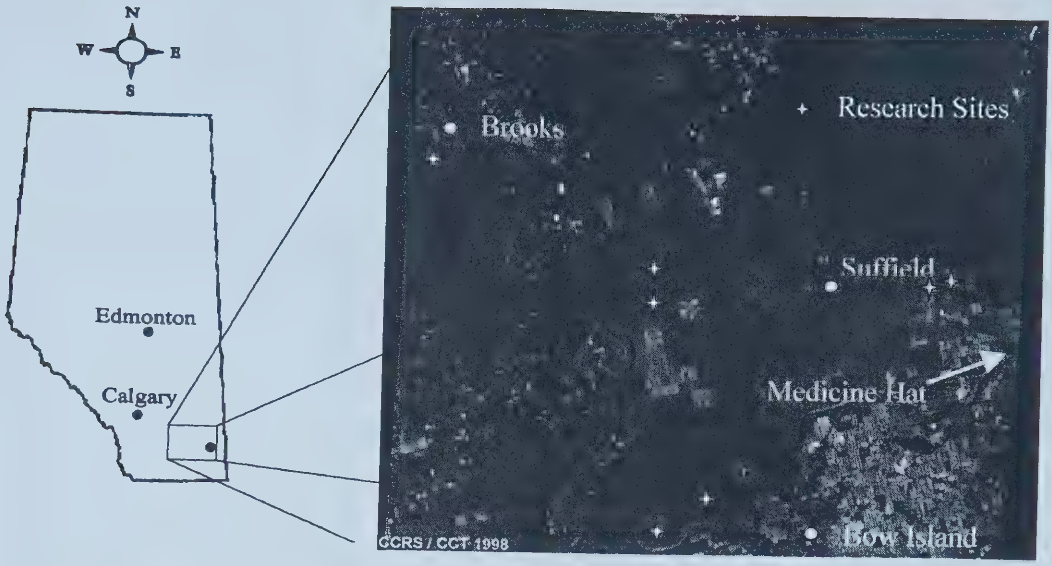

Figure A-1: Research site locations in southeastern Alberta.

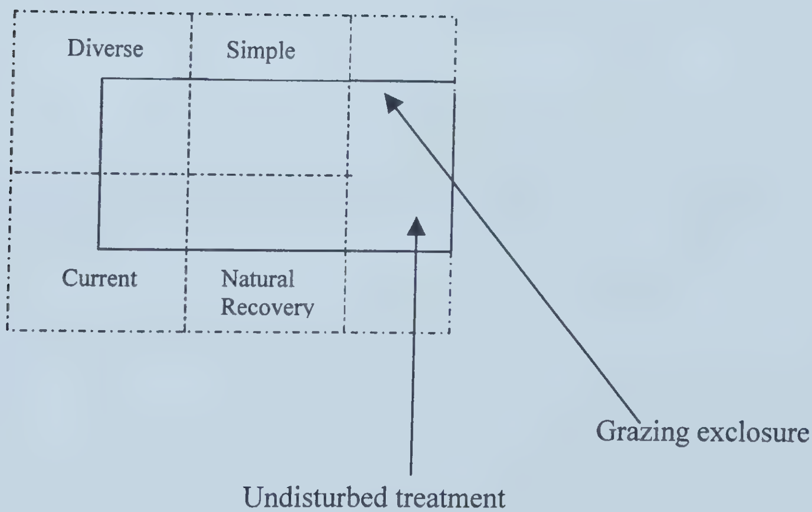

Figure A-2: Treatment layout for strip plot design.

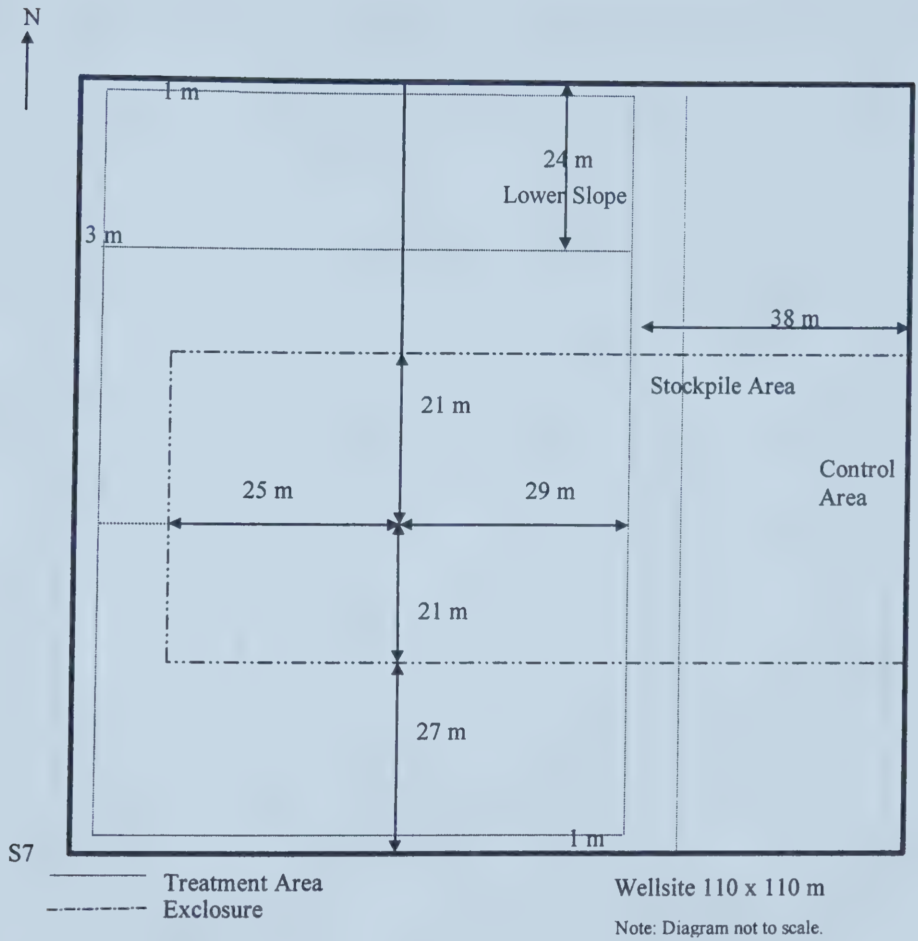

Figure A-3: Site diagram for Intensity East (#100 3-29-14-7 W4).

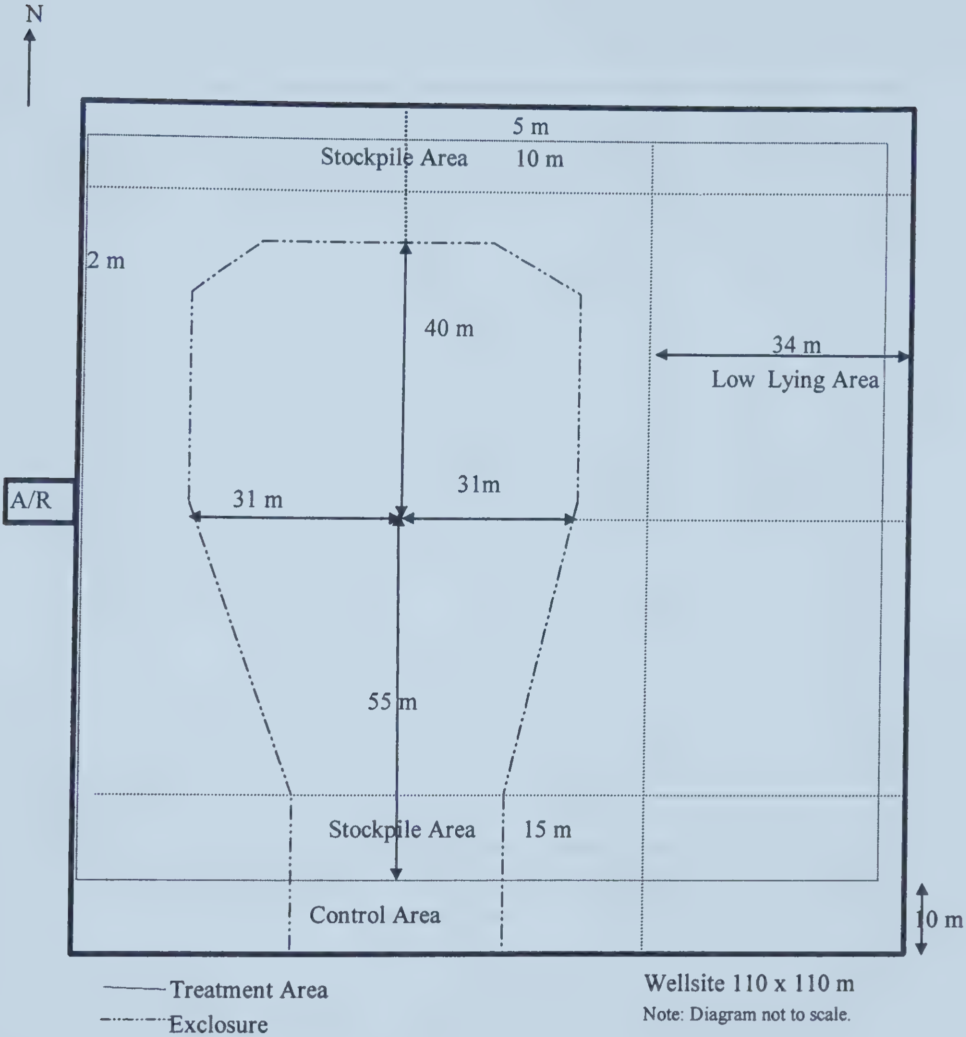

Figure A-4: Site diagram for Intensity West (#102 3-29-14-7 W4).

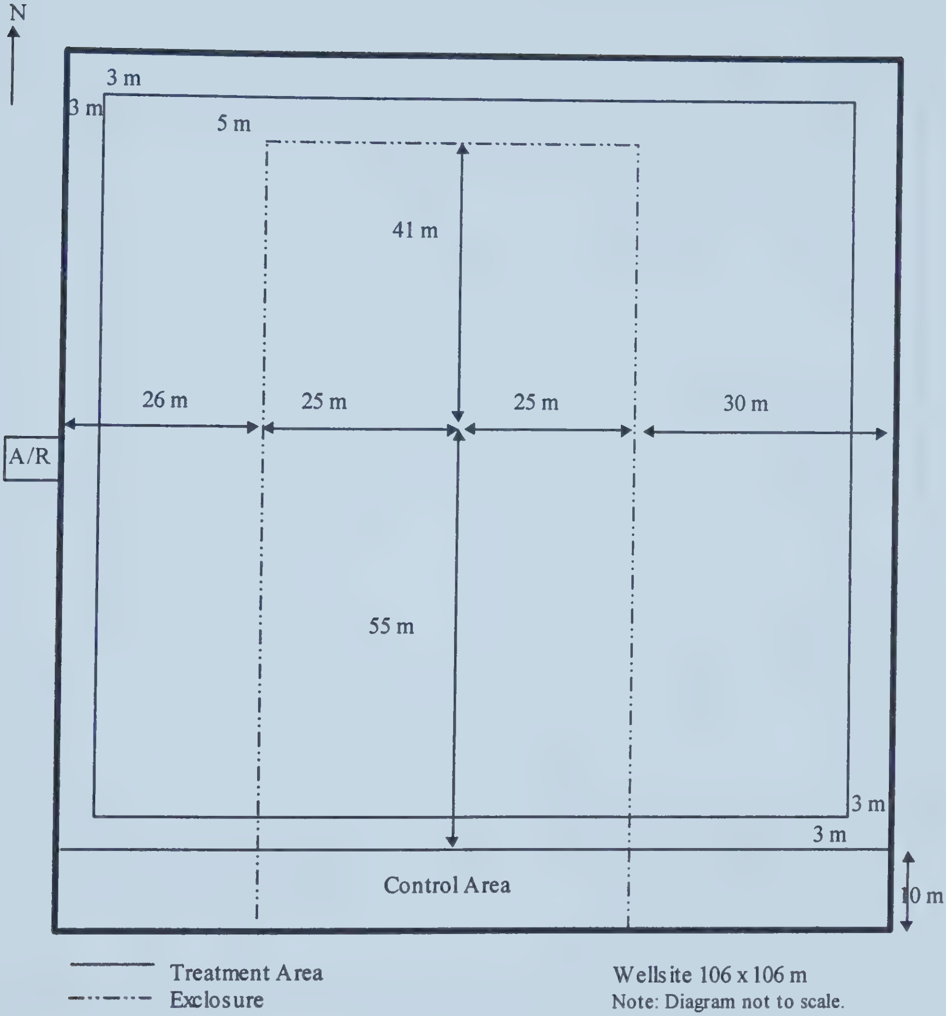

Figure A-5: Site diagram for Sceptre (13-18-11-12 W4).

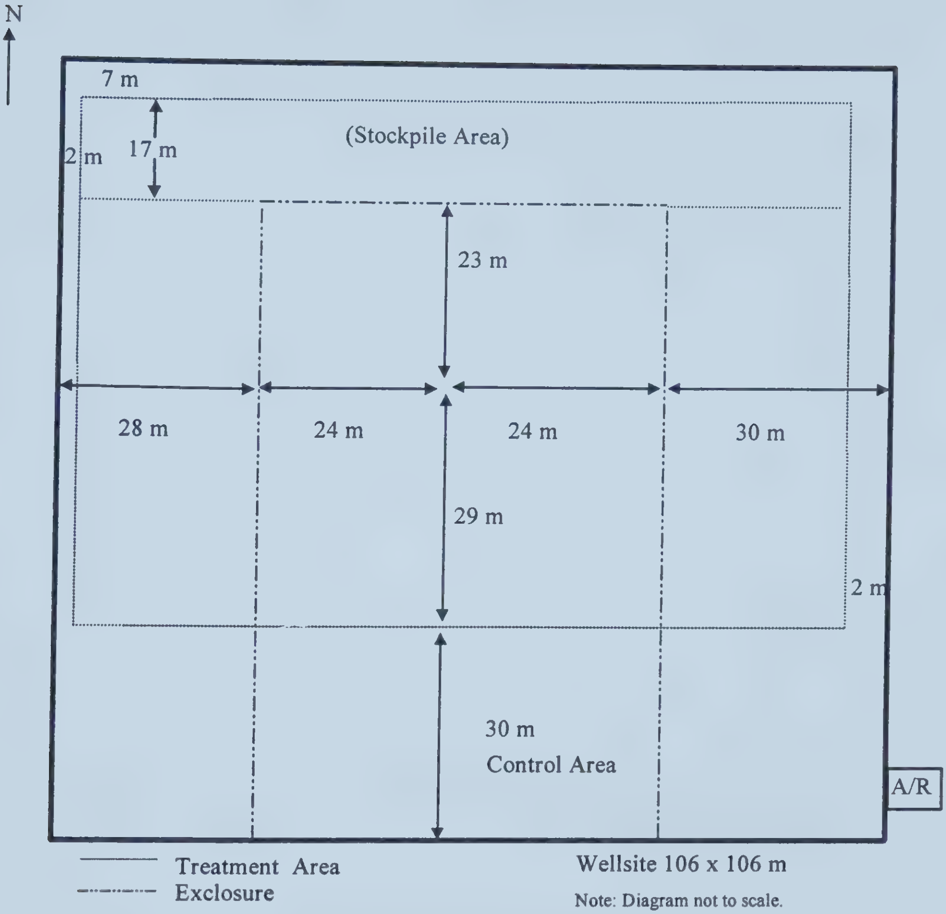

Figure A-6: Site diagram for Startech (1-35-11-12 W4).

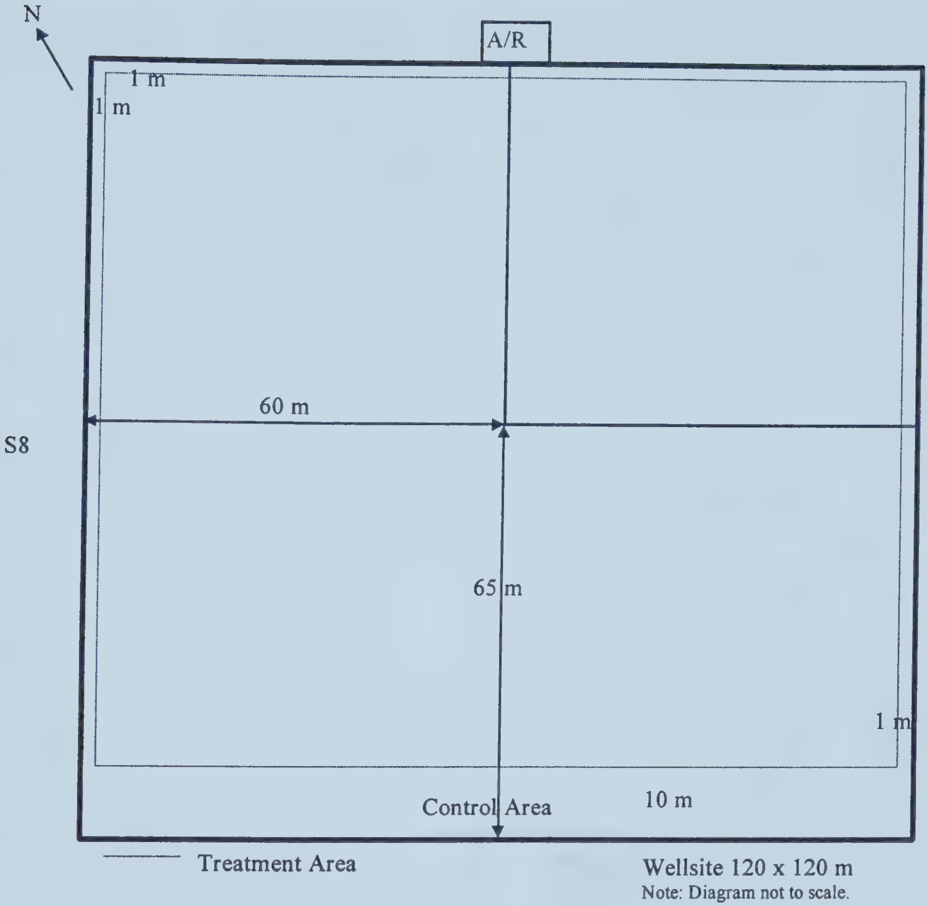

Figure A-7: Site diagram for PCP North (7-32-15-11 W4).

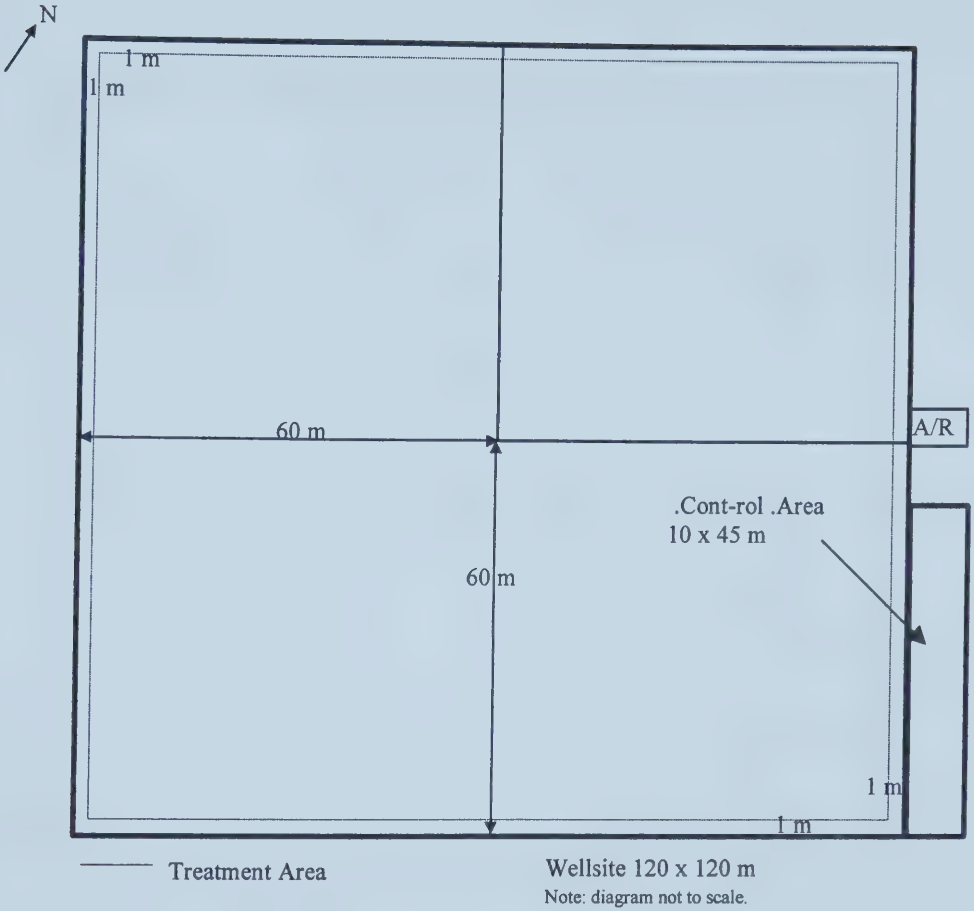

Figure A-8: Site diagram for PCP South (7-20-15-11 W4).

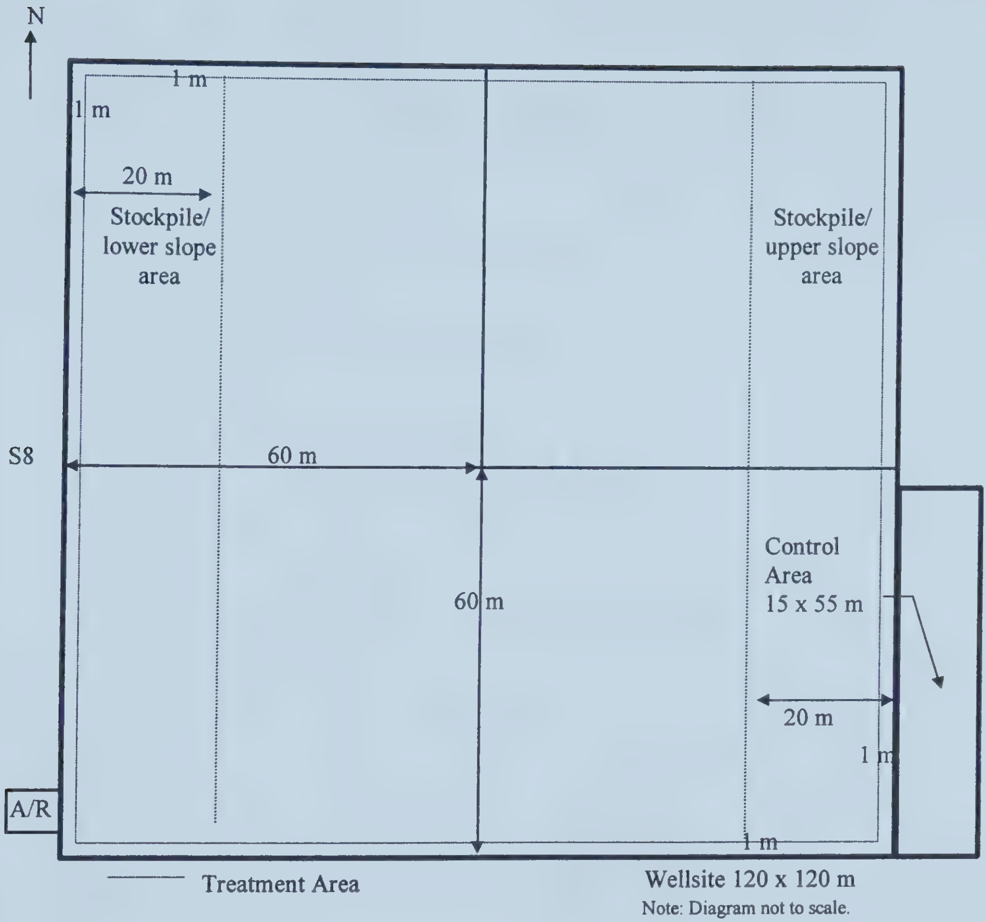

Figure A-9: Site diagram for PCP West (15-3-18-15 W4).

Appendix II – Latin and Common Names of Plant Species

Agropyron desertorum (Fisch. ex Link) Shultz. (crested wheatgrass)
Achillea millefolium L. (yarrow)
Agropyron dasystachyum (Hook.) Scribn. (northern wheatgrass)
Agropyron smithii (Rydb.) (western wheatgrass)
Agropyron trachycaulum (Link) Malte (slender wheatgrass)
Amaranthus spp. L. (amaranth)
Ambrosia trifida L. (great ragweed)
Aster ericoides L. (tufted white prairie aster)
Astragalus striatus Nutt. (milk vetch)
Avena fatua L. (wild oats)
Bouteloua gracilis (HBK) Lag. (blue grama)
Carex spp. (sedges)
Chenopodium album L. (lamb's-quarters)
Danthonia spicata (L.) Beauv. (poverty oat grass)
Descurania spp. Webb & Berthelot (tansy mustard)
Elymus canadensis L. (canada wild rye)
Elymus junceus Fisch. (russian wildrye)
Gaillardia aristata Pursh (gaillardia)
Gutierrezia sarothrae (Pursh) Dunal (broom snakeweed)
Hedysarum sulphurescens Rybd. (hedysarum)
Hordeum jubatum L. (foxtail barley)
Kochia scoparia (L.) Schrad. (kochia weed)
Koeleria macrantha (Ledeb.) J.A. Schultes f. (june grass)
Lappula spp. Moench (beggars-ticks)
Lepidium densiflorum Schrad. (common peppergrass)
Oryzopsis hymenoides (R. & S.) Ricker (indian rice grass)
Petalostemon candidum (Willd.) Michx. (prairie clover)
*Petalostemon purpureu*m (Vent.) Rybd. (purple prairie clover)
Poa spp. L. (blue grass)
Ratibida columnifera (Nutt.) Wooten & Standl. (prairie cone flower)
Salsola kali L. (Russian thistle)
Schizachrium scoparium (Michx.) Nees (little bluestem)
Selaginella densa Rydb. (little club moss)
Setaria glauca (L.) Beauv. (yellow foxtail)
Setaria viridis (L.) Beauv. (green foxtail)
Sitanion hystrix (Nutt.) J.G. Smith (squirreltail)
Solidago canadensis L. (canada goldenrod)
Sonchus spp. (sow thistle)
Stipa comata Trin & Rupr. (needle and thread)
Stipa viridula Trin. (green needle grass)
Thermopsis rhombifolia R.Br. (golden bean)
Vicia americana Muhl. (wild vetch)

APPENDIX III – EXAMPLE OF ANOVA TABLE USED FOR STRIP PLOT DESIGN

Table A-5: ANOVA table used for strip plot design.

Source	Degrees of Freedom
Block	3
Revegetation treatment (a)	4
Grazing treatment (b)	1
Grazing by revegetation interaction	4
Revegetation by block interaction (error A)	12
Grazing by block interaction (error B)	3
Residual error (error C)	12